UNIVERSITY OF TASMANIA

Demography of Shy and White-capped albatrosses: Conservation Implications

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Dedication

To Steve Wilson and Stephen Marchant, who taught me about the study of birds

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Thesis Abstract

Shy and white-capped albatrosses, *Thalassarche cauta* and *T. steadi* respectively, are closely related and phenotypically similar seabird species. Shy albatrosses breed in Australia on three islands around Tasmania, whereas white-capped albatrosses breed on islands in the Auckland and Antipodes Islands groups in New Zealand's subantarctic. Humans have impacted shy albatrosses for over a century, with at least one population devastated by feather and egg collectors in the early 1900s. Both species are also caught and killed as bycatch in fisheries across a wide range. The impact of this threat alone on these species may well be unsustainable.

Here I have adopted two approaches to prepare a current conservation assessment of the both shy and white-capped albatrosses. Both approaches have been used independently in studies to assess the impacts of fisheries related mortality on other seabird species, but rarely have both been implemented simultaneously. First, I reviewed the levels of effort in fisheries known to kill both species and developed an estimate of the global bycatch level. Second, I developed and fitted population models for both species to evaluate the impact of bycatch on population growth. I also undertook annual population censuses of white-capped albatrosses at three sites in the Auckland Islands (where 99% of the population breed), from 2006 to 2013, to estimate population size and track population trends.

I complemented these analyses with at-sea experiments to test the efficacy of a mitigation method known as the Smart Tuna Hook (STH). This method employs a specially designed shield that disarms the hook once it has been baited, making it difficult for any seabird to be hooked. The shield is released within 15 minutes of the hook being immersed in salt water. The experimental work was conducted on tuna longline vessels fishing out of Cape Town, South Africa, and involved a direct comparison of the Smart Tuna Hook and conventional pelagic hooks in tuna fishing operations.

The analyses of global fishing effort and fisheries bycatch rates indicate that over 8 500 shy and white-capped albatrosses may be killed annually. Trawl fisheries

were responsible for 75% of all estimated mortality, with longline fisheries accounting for 25%. Most birds were killed in South African, Namibian and New Zealand fisheries. As most adult shy albatrosses are comparatively sedentary and rarely found outside Australian waters, it is primarily juvenile shy albatrosses that regularly encounter fishing fleets known to kill large numbers of albatrosses. In contrast, throughout most of their range both juvenile and adult white-capped albatrosses are exposed to fisheries that collectively kill many thousands of these birds each year.

The Auckland Island censuses estimated the mean number of annual breeding pairs to be 90 141, with annual estimates ranging from 73 838 to 116 025 pairs. Trend analysis using regression splines showed no clear evidence for monotonic decline, providing insufficient evidence to reject the null hypothesis of no trend in the total population. Trend analysis using Program TRIM, currently used by the Agreement on the Conservation of Albatrosses and Petrels to assess albatross population trends, indicated an average growth rate of -3.16% per year, assessed by TRIM as moderate decline. However, a simple linear trend analysis as performed by TRIM is not well suited to a data set with high inter-annual variability. I therefore concluded that the population trend is uncertain.

Population models developed for both shy and white-capped albatrosses indicated that the levels of estimated global fisheries bycatch is unsustainable for both species, and particularly for white-capped albatrosses. However, as the observed population trend for both species over the last 10 years has not shown the rate of decline predicted by modelling, it is likely that the bycatch estimates for both species have been over-estimated. The Potential Biological Removal level calculated for white-capped albatross and used in current risk prioritisation is also likely to be unsustainable. Application of a PBR based on a low recovery factor (FR = 0.1 or FR = 0.2) would be appropriate for both species.

While considerable progress has been made in mitigating bycatch in trawl and demersal longline fisheries, proven seabird avoidance measures in pelagic fisheries require substantial improvement. My tests of the Smart Tuna Hook showed that bycatch could be reduced by between 81.8% – 91.4%. Importantly, there was no detectable detrimental effect on fish catch for any commercial species. The Smart Tuna Hook provided a significant deterrent to seabirds attacking baits, and offers a feasible option for pelagic fishers to significantly reduce seabird bycatch.

The bycatch of shy and white-capped albatrosses occurs over the entire range of both species and at levels that are impacting population growth. Reducing bycatch in fisheries poses significant challenges for gear technologists and fisheries managers. Finding solutions requires a mix of legislative and political measures to facilitate industry engagement and provide incentives for action, combined with sound science to define problems and provide robust assessments of the impact of bycatch at a species and population level, and to ensure development and implementation of effective mitigation measures.

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General Introduction

Seabirds are killed in a range of fisheries throughout the world, and there is evidence that fisheries-related mortality is responsible for population decreases in many species, particularly the albatrosses and petrels (Families Diomedeidae and Procellariidae) (Alexander et al. 1997; Croxall 1998; Gales 1998; Baker et al. 2002). This threat to seabirds has been particularly well documented for longline fisheries, where birds drown after being accidentally caught while scavenging on baited hooks set for target pelagic and demersal fish. Mortality of seabirds associated with trawl, gillnet and purse-seine fisheries is less well documented but is increasingly recognised, especially in trawl fisheries where seabirds can get struck by the warp lines and drown, collide with other vessel cables and be killed or injured, or become entangled in the mesh of nets at the sea surface. In gillnet and purse-seine fisheries, seabirds can become entangled in the mesh of nets and drown, either accidentally, or as a result of active diving / feeding behaviour of certain species.

Seabird mortality arising from fisheries interactions have been linked to population declines in many species (Weimerskirch and Jouventin 1987; Croxall et al. 1990; Weimerskirch et al. 1997; Gales 1998; Weimerskirch and Jouventin 1998; Tuck et al. 2001; Nel et al. 2002). While there has been an attempt in recent years to address these concerns in some fisheries (e.g. Environment Australia 1998; Baker and Finley 2010), bycatch information is limited for those species that are difficult to identify using morphometric or plumage characteristics (Double et al. 2003; Abbott et al. 2006). This information is essential in assessing the impact of fisheries-related mortality on individual seabird species.

Shy and white-capped albatrosses, *Thalassarche cauta* and *T. steadi* respectively, are closely related and phenotyically similar species. Once considered to be a single species, genetic work by Abbott and Double (2003) provided strong evidence that they are distinct species. Shy albatrosses breed in Australia on three islands around Tasmania, whereas white-capped albatrosses breed on islands in the Auckland and Antipodes Islands group in New Zealand’s sub-Antarctic. Shy albatrosses have been impacted by human impacts for over a century. Formerly abundant, at least one population was nearly exterminated by feather and egg collectors at the beginning of the last century (Johnstone et al. 1975) and, while human predation no longer constitutes a threat, disease appears to be slowing population recovery at one site (Johnstone et al. 1975; Woods 2004). White-capped albatross have been better protected on their breeding grounds by spatial isolation, but are known to have been eaten by shipwrecked sailors on occasions (Peat 2006) and one colony is known to be predated by pigs following their introduction to Auckland Island in 1807, a threat which still continues unabated today (Tickell 2000 In addition to these historical and contemporary population pressures, both species are known to suffer fisheries-related bycatch mortality across a wide range, including in Australian, New Zealand and South African waters (Bartle 1991; Gales 1998; Ryan et al. 2002). Given the spatial extent and potential impact of this threat, like many of the other petrels and albatrosses, its impact on shy and white-capped albatrosses may well be unsustainable.

Assessing the impact of fishery-related mortality is usually addressed by estimating the number of birds killed by a particular fishery. Such assessments rely on the use of fisheries effort and observer data, combined with knowledge of the geographic foraging distributions of a species, to evaluate the spatial and temporal overlap with a fishery. These data are often limited or hard to access, thus constraining effective conservation assessment of the threat to a species by bycatch.

An alternative approach to assessing the impact of fisheries is to estimate vital demographic rates (survival, fecundity, immigration and emigration) and the relative contribution they make to population growth rate, to assess trends in populations. If bycatch levels and other forms of mortality are unsustainable, populations will exhibit negative population growth. This latter approach is dependent upon the available of demographic data, ideally collected over multiple consecutive years to allow identification of a population trend.

In this thesis I adopt both these approaches to prepare a current conservation assessment of the both shy and white-capped albatrosses. First, current levels of effort in fisheries known to kill both species will be used to evaluate the scale of the problem on a fishery-by-fishery basis, leading to an estimate of the global bycatch level. To my knowledge, while such assessments have been attempted for individual fisheries, no assessment has ever been attempted for all fisheries likely to impact a species of seabird. Secondly, published literature and analysis of existing long-term demographic data sets for both species will be used to estimate survival and breeding success parameters. These parameters will be incorporated into a Population Viability Analysis (PVA) to assess the conservation implications of fishing-related mortality and other threats, and provide guidance for future management of both species. PVA analysis will also consider potential biological removal (PBR) levels (Wade 1998) for both shy and white-capped albatrosses Finally, I will consider global efforts to conserve albatrosses and petrels, specifically through the development and implementation of mitigation measures, and examine the potential for a new mitigation tool, the Smart Tuna Hook, to reduce bycatch of seabirds generally, and shy and white-capped albatrosses in particular.

My thesis is organised into four chapters. Each chapter is designed to stand-alone and includes a thorough introduction to the specific topic and a review of relevant literature. In consequence, I have kept the background information about each species to a succinct minimum in this general Introduction. All chapters are written in a style that is suitable for publication. One has already been published and others have been submitted, or are ready for submission. The thesis begins with a review of global bycatch in shy type albatrosses, and is then followed by a population assessment of the white-capped albatross, a necessary tool to evaluating the impact of bycatch on a species. Such work was not necessary for the shy albatross and has already been done by others (Alderman et al. 2011). The third chapter develops a population viability assessment for both shy and white-capped albatrosses, and the final chapter evaluates a bycatch mitigation device that has the potential to significantly reduce the incidental of shy-type albatrosses and other seabirds. A brief outline of my thesis is as follows:

- In Chapter 1 I assessed the likely impact of bycatch mortality on shy and white-capped albatrosses throughout the Southern Oceans on a fishery-by-fishery basis, based on knowledge of their distribution and the extent of the associated fisheries. To do this I examined current levels of effort in fisheries known to incidentally kill shy-type albatrosses, and subsequently evaluated the degree to which they overlap with each of the two species.

- In Chapter 2 I estimates the size of the white-capped albatross population through aerial photography and assessed population status and trend from 2006 to 2013. This information is critical for informing management on how bycatch levels are affecting population dynamics.

- In Chapter 3 I assessed the impact of fisheries on the viability of both shy and white-capped albatross populations by conducting a Population Viability Analysis for both species. I evaluated current levels of fisheries-related mortality, as well as estimating a potential biological removal value for both species.

- In Chapter 4 I compared the Smart Tuna Hook, a device designed to restrict the access of seabirds to baited hooks on pelagic longline fishing gear, with standard fishing equipment, to assess the efficacy of this measure in reducing seabird bycatch whilst still maintaining catch of target species. . Due to high mortality levels longline fishing has been identified as a major threat affecting many seabird species, and finding ways of reducing the impact of this threat is essential for seabird conservation.

- I conclude the thesis with a summary of the primary findings from each chapter and discuss them in the context of their implications for both species. I also synthesise these findings in a discussion of realistic and achievable measures that can be implemented to assist in the conservation and long-term management of seabirds generally.

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Chapter 1

A global assessment of the impact of fisheries-related mortality on shy and white-capped albatrosses: conservation implications

Baker, G.B., Double, M.C., Gales, R., Tuck, G.N., Abbott, C.L., Ryan, P.G., Petersen, S.L., Robertson, C.J.R. and Alderman, R. 2007. A global assessment of the impact of fisheries-related mortality on shy and white-capped albatrosses: conservation implications. Biological Conservation 137: 319—333.

Abstract

Hundreds of thousands of seabirds are killed each year as a result of interacting with longline and trawl fishing operations, and the severity of the impact varies regionally. Shy and white-capped albatrosses, *Thalassarche cauta* and *T. steadi* respectively, are phenotypically similar species known to be incidentally killed by fishing operations. The magnitude of this mortality has not previously been assessed across their range. Here I examine recent effort and bycatch rates in fisheries known to incidentally kill these species and qualitatively evaluate the level of impact of each fishery. Results indicate that over 8 500 of these albatrosses may be killed annually, although the reliability of this estimate is low due to the paucity of comprehensive observer data in most fisheries. Of the estimated deaths of all seabird species in the fisheries assessed, trawl and longline fisheries killed birds in approximately equal proportions, but when the mortality levels of shy-type albatrosses were examined, trawl fisheries were responsible for 75% of all deaths. Data suggest most of these birds were killed in South African, Namibian and New Zealand demersal trawl fisheries and the South Africa Pelagic Longline Fishery. Because most adult shy albatrosses are comparatively sedentary and rarely found outside Australian waters, it is primarily juvenile shy albatrosses that regularly encounter fishing fleets known to kill large numbers of albatrosses. In contrast, throughout most of their range juvenile and adult white-capped albatrosses are exposed to fisheries that collectively kill many thousands of these albatrosses each year. These data emphasise the urgent need for robust assessments of the impact of bycatch at a species and population level, and the urgent implementation of effective mitigation measures.

Introduction

Seabird deaths arising from fisheries interactions have been linked to population declines in many species (Weimerskirch and Jouventin 1987; Croxall et al. 1990; de la Mare and Kerry 1994; Prince et al. 1994; Weimerskirch et al. 1997; Gales 1998; Weimerskirch and Jouventin 1998; Tuck et al. 2001; Nel et al. 2002). While there has been an attempt in recent years to address these concerns in some fisheries (e.g. Environment Australia, 1998; CCAMLR, 2004), bycatch information is limited for those species that are difficult to identify using morphometric or plumage characteristics. This information is essential in assessments of the impact of fisheries-related mortality on individual seabird species.

Shy and white-capped albatrosses, *Thalassarche cauta* and *T. steadi* respectively, are closely-related and phenotyically similar species (Abbott and Double 2003 a, b; Double et al. 2003). Shy albatrosses breed in Australia on three islands around Tasmania, whereas white-capped albatrosses breed on islands in the Auckland and Antipodes Islands group in New Zealand's subantarctic. Their global population sizes are estimated at 12 000 and 75 000 annual breeding pairs, respectively (Gales 1998[1]). These species were classified as Vulnerable by Croxall and Gales (1998)

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[1] This Chapter and subsequent published paper was written in 2007 and the population estimates provided here have been subsequently updated following the work of Alderman et al. (2011) and my work in Chapter 2. For consistency with my published paper I have not changed these estimates in this Chapter. The citation for Alderman et al. (2007) and my paper follow:
Alderman, R.L., Gales, R., Tuck, G.N., Lebreton, J.D. 2011. Global population status of shy albatross and an assessment of colony-specific trends and drivers. Wildlife Research 38, 672-686.
Baker, G.B., Double, M.C., Gales, R., Tuck, G.N., Abbott, C.L., Ryan, P.G., Petersen, S.L., Robertson, C.J.R., Alderman, R. 2007. A global assessment of the impact of fisheries-related mortality on shy and white-capped albatrosses: conservation implications. Biological Conservation 137, 319–333.

using IUCN criteria but are listed as Near Threatened by Birdlife International, who have only recently recognised the white-capped albatross as a separate species. Both species are known to suffer fisheries-related bycatch mortality across a wide spatial scale, including in Australian (Brothers 1991; Gales 1998), New Zealand (Murray et al 1993; Robertson et al. 2004), and South African (Ryan et al. 2002) waters.

Previously, it has not been possible to assess the extent or scale of impact of bycatch mortality on shy and white-capped albatrosses, in part due to an inability to identify bycatch shy-type albatross carcasses to species level. However, Abbott et al. (2006) recently used molecular species assignment methods to distinguish shy-type albatross carcasses obtained from fisheries bycatch in Australia, New Zealand, and South Africa waters. This provided novel information on the geographic distributions of these species, and provided an index of relative abundance of each species across continental shelf regions known to be heavily exploited by shy-type albatrosses (Bartle 1991; Gales 1998; Ryan et al. 2002). They found that shy and white-capped albatrosses had vastly different at-sea distributions. Adult shy albatrosses were only detected in Australian waters, largely confirming the results of banding and tracking studies that indicated juveniles forage widely in the waters off southern Australia whereas adult shy albatrosses remain close to their breeding colonies throughout the year (Brothers et al. 1998; Hedd et al. 2001; Hedd and Gales 2005). The banding and tracking studies also showed that juvenile shy albatrosses can reach South African and New Zealand waters (Brothers et al. 1998), although they were not found among the bycatch samples from these areas (Abbott et al. 2006).

In contrast, Abbott et al. (2006) found that both juvenile and adult white-capped albatrosses were recovered from New Zealand, southern Australian and South African waters. Besides this study, there have been no detailed study of the at-sea distribution of white-capped albatrosses, but two banded birds have been recovered in Namibia and South Africa (Robertson et al. 2003b 2006).

Because of the differences in the pelagic distribution it is likely shy and white-capped albatrosses face different levels of threat from interactions with trawl and longline fisheries. The primary aim of this study was to assess the likely impact of bycatch mortality on these species throughout the Southern Oceans on a fishery-by-fishery basis, based on improved understanding of their distribution. To do this I examined current levels of effort in fisheries known to incidentally kill shy-type albatrosses, and subsequently evaluated the degree to which they overlap with each of the two species. This is the first study to estimate the impact of fisheries bycatch mortality for each species separately, and will assist in determining the conservation implications of fisheries bycatch mortality for both species across their ranges.

Methods

Nomenclature

In the taxonomic revision of albatrosses suggested by Robertson and Nunn (1998), the shy albatross complex was split into four species: Salvin's albatross (*Thalassarche salvini*); Chatham albatross (*T. eremita*); white-capped albatross (*T. steadi*) and the shy albatross (*T. cauta*). The recognition of the latter two taxa as separate species was supported by later morphometric, phylogenetic and population genetic studies (Abbott and Double 2003a, b; Double et al. 2003). I therefore use the names 'shy albatross' and 'white-capped albatross' and refer to them as species. The term 'shy-type' is used to refer to these two taxa collectively.

In this paper I refer to all birds yet to acquire the body characteristics of full adults as 'juvenile'. By a country's 'waters' I am referring to its Exclusive Economic Zone (EEZ), which usually extends 200 nautical miles from the country's coastline.

Breeding sites

Shy albatrosses are endemic to Australia, breeding only on three islands around Tasmania: Albatross Island (5 000 pairs); the Mewstone (7 000 pairs) and Pedra Branca (200 pairs) (Fig.1). The total population is estimated to be 55 000 – 60 000 individuals (Gales 1998). Breeding is annual; females lay a single egg in September each year, with chicks hatching in December and fledging in April (Gales 1993; Hedd and Gales 2005; Abbott et al. 2006). After departing their colonies, young birds spend at least two years at sea before returning to their natal colony and do not start breeding until they are at least five years old (Rosemary Gales unpublished data).

White-capped albatrosses are endemic to New Zealand, breeding on Disappointment Island (72 000 pairs), Adams Island (100 pairs) and Auckland Island (3 000 pairs) in the Auckland Island group, and Bollons Island (50 – 100 pairs) in the Antipodes Island Group (Fig.1). The population is estimated to comprise about 350 000 – 375 000 individuals (Gales 1998; Abbott et al. 2006). The breeding frequency and season for this species are poorly known. Egg-laying is reported to commence in November, with chicks hatching in February and fledging in August (Robertson 1985).

At-sea distribution

Because shy and white-capped albatrosses are difficult to distinguish at-sea, knowledge of their distribution is based largely on satellite tracking, banding and molecular identification of bycatch specimens.

Satellite tracking and studies of colony attendance have shown that adult shy albatrosses are remarkably sedentary (Brothers et al. 1997; Brothers et al. 1998; Hedd et al. 2001; Hedd and Gales 2005). When breeding, shy albatrosses feed mainly during daylight hours over the continental shelf within 200km of their breeding colonies (Brothers et al. 1998; Hedd et al. 2001). After fledging their chicks, adults from Albatross Island spend just nine weeks foraging off southern Australia before returning to attend their nest sites (Hedd and Gales 2005).

A study of shy albatrosses that applied some 21 000 bands between 1960 and 1995 (mostly to fledgling shy albatrosses) indicated that juveniles (<seven years old) from Albatross Island and Mewstone have different at-sea distributions (Brothers et al. 1997). Of the 15 927 birds banded on Albatross Island the 120 recoveries (up to 1995) of juvenile birds were all made in Australia. In contrast, of the 5 009 birds banded on Mewstone, 21 bands were recovered on juveniles, 57% in Australia, 38% in South Africa and 5% in New Zealand. None of the 366 birds banded on Pedra Branca have been recovered (Brothers et al. 1997). Overall nearly 63% of all juveniles were recovered in Tasmanian and Victorian waters and 20% were recovered in South and Western Australia. Since the study by Brothers et al. (1997), two juvenile shy albatrosses, banded as fledglings on Albatross Island, have been recovered as longline bycatch off South Africa and in the Tasman Sea (Rosemary Gales unpublished data). Whilst the movements of young shy albatrosses remains poorly known, current knowledge suggests that juvenile shy albatrosses forage mainly off southern and western Australia but some birds, predominantly from the Mewstone population, traverse the Indian Ocean to forage in waters off South Africa.

There have been no published satellite tracking or banding studies of white-capped albatrosses so current knowledge of their at-sea distribution is based largely on the study by Abbott et al. (2006) who, by DNA-based identification, found that all bycatch samples of shy-type albatrosses from New Zealand (97.5% adults; N=80) and South Africa (N=25) were white-capped albatrosses. Further analyses using the 'SNP test' of Abbott et al. (2006) found that of a sample of 254 shy-type albatrosses killed in South African fishing operations, 241 (94.9%) were white-capped albatrosses, and 13 (5.1%) shy albatrosses (MCD and PGR, unpublished data). Abbott et al. (2006) reported that of those samples identified to be from shy albatrosses by the 'SNP test' approximately 3% were in fact from white-capped albatrosses; no error has been associated with samples from white-capped albatrosses. Given this low error rate I estimate that 5% of the shy-type albatrosses killed in South African waters are shy albatrosses.

Ryan et al. (2002; Abbott et al. 2006) reported that approximately 44% of shy-type albatrosses caught in South Africa waters were adults. Of 93 shy-type albatrosses caught off Tasmania, 34% were identified as white-capped albatross of which 37% were adults (Abbott et al. 2006). These data suggest that adult white-capped albatrosses are less sedentary than shy albatrosses and that both adult and juvenile white-capped albatrosses occur in approximately equal frequencies in Australian and South African waters. Also white-capped albatrosses dominate the shy-type assemblage in New Zealand and South Africa (Abbott et al. 2006).

Shy-type albatrosses have also been recorded in both the south Atlantic and eastern Pacific (Tickell 2000; Phalan et al. 2004), but sightings are uncommon and these areas are not considered to be part of their usual distribution (Tickell 2000; Robertson et al. 2003b). Indeed, none of the many surveys of seabird bycatch in South American waters have, as yet, reported shy-type albatrosses among the birds caught (e.g. Neves and Olmos 1998; Stagi et al. 1998; Reid et al. 2004; Sullivan et al. 2004; Moreno et al. 2006; Sullivan et al. 2006).

Fisheries effort and assessment

I have followed the general approach of Tuck et al. (2003) to describe recent temporal trends in effort in fisheries that are within the distribution of, and have been reported to kill, shy-type albatrosses. These fisheries are listed in Table 1.

Fisheries effort often varies considerably over time in response to changing economic and biological factors so I have acquired at least five years of fisheries data (post 1995) and attempted to obtain the most recent publicly available data. Effort data were obtained on a spatial scale of 5^o X 5^o for each fishery. Outside of South African and Namibian EEZs I only included effort south of latitude 30^oS and between longitudes 0^o to 160^oW (Fig.1). Based on current understanding of the distribution of shy and white capped albatrosses, this excludes effort not thought to significantly overlap with the distribution of these species (Abbott et al. 2006). Recently, within South African and Namibian EEZs, shy-type albatrosses have been observed and satellite-tracked north of 30^0S (Samantha Petersen unpublished data). Data were analysed annually, and further sectioned into four three-monthly seasons (January – March, April – June, July – September, October – December). The total area of interest was divided into five areas: Areas 1 to 3 represent the major foraging zones of juveniles of both species (but possibly not all populations of shy albatrosses) and the foraging area of non-breeding white-capped albatrosses (Brothers et al. 1997; Robertson et al. 2003a, R. Gales and R. Alderman unpublished data); Areas 4 and 5 encompass the known or assumed core foraging areas of breeding shy (Brothers et al. 1998, R. Gales and R. Alderman unpublished data) and white-capped albatrosses (Robertson et al. 2003a), respectively (Fig.1).

Data were obtained from international fishery commissions (Indian Ocean Tuna Commission – IOTC; Secretariat of the Pacific Community – SPC; International Commission for the Conservation of Atlantic Tunas – ICCAT; Commission for the Conservation of Southern Bluefin Tuna – CCSBT;) and national fisheries agencies (Australian Fisheries Management Authority; Primary Industries Research Victoria, Australia; New Zealand Ministry of Fisheries; and South African Department of Marine and Coastal Management).

Because fisheries observer data were generally poor for most fisheries (see below) I used the available data to estimate the total annual mortality of seabirds, and the bycatch component that comprised shy and white-capped albatrosses. I then qualitatively categorised the impact of each fishery on seabirds as a two stage process — first for all seabirds, and then specifically for shy-type albatrosses. I arbitrarily categorised the impact for both groups as ‘low’, ‘medium’, ‘high’ or ‘very high’ if it was estimated that <100, 100 to 499, 500 to 999 or >1 000 birds were killed in the fishery each year respectively. Where no observer data were available I inferred the potential impact on both shy and white-capped albatrosses based on the known distribution of these species and the type of operation used by the fishery.

When observer coverage is low or not representative, extrapolations are potentially inaccurate and misleading (Uhlmann et al. 2005). Agnew (2001) suggested that the level of observer coverage needed to accurately estimate bycatch levels in longline fisheries is 20% of all hooks set. Based on this recommendation I assigned a reliability indicator to each fishery assessment: 'low' when less than 10% of hooks were observed; 'medium' when 10 to 20% of hooks were observed and high if >20% of hooks were observed. I adopted a similar regime for trawl fisheries based on percentage of trawls or trawl-hours observed.

Results

South African Pelagic Longline Fishery

Between 1998 and 2000 Japanese and Taiwanese vessels, primarily targeting tuna, (*Thunnus* spp), accounted for 96% of the c. 12 million hooks set annually (Fig.2 (a)). In 2002, all foreign licences were revoked but by 2005 the 43 licensed vessels include 15 foreign, mostly Korean, flagged vessels (Petersen et al. 2006). Domestic vessels primarily target broadbill swordfish (*Xiphias gladius*) whereas foreign vessels usually set their gear deeper and target albacore (*Thunnus alalunga*), yellowfin (*T. albacares*) and bigeye tuna (*T. obesus*) (Cooper and Ryan 2002). Effort is usually concentrated along the edge of the continental shelf although some vessels fish farther offshore in the Atlantic and Indian Oceans, outside the South African EEZ (Cooper and Ryan 2002).

Observer data from 1998 to 2000 reported 2.6 birds killed per 1 000 hooks on Japanese vessels (1% effort observed) and 0.8 birds killed per 1 000 hooks on domestic vessels (17% effort observed) (Ryan et al. 2002). Of 101 seabirds retained for identification from Japanese vessels, 37% were identified as shy-type albatrosses. Ryan et al. (2002) suggest that given the level of effort during this period (~11.5 million hooks annually) and fleet composition (~4% of effort by domestic vessels) between 19 000 and 30 000 seabirds were killed annually in this fishery, of which between 7 000 and 11 000 were shy-type albatrosses.

Cooper and Ryan (2002) later reported a bycatch rate on domestic vessels of 0.34 birds killed per 1 000 hooks. This was based on observing the deployment of 294 000 hooks on 36 trips between 1998 and 2001, 11 of which were previously included in the dataset presented by Ryan et al. 2002. Shy-type albatrosses comprised 23% of all birds caught.

In 2004 and 2005, joint-venture vessels were estimated to catch 0.6 seabirds per 1 000 hooks (80% observer coverage) whereas between 2000 and 2004 domestic boats were estimated to catch 0.2 seabirds per 1 000 hooks (11% observer coverage) (Petersen 2004). Of birds retained for identification 28% were shy-type albatrosses (Petersen 2004). In 2005, c. 3.9 million hooks were set (Fig.2 (a)) of which 80% were set by foreign vessels operating under joint-venture agreements. These data suggest that about 2 000 seabirds were killed in 2005, of which between 500 and 600 were shy-type albatrosses. I therefore assess this to be a ‘very high impact’ fishery for seabirds overall, and a ‘high’ impact fishery for shy-type albatrosses. Observer coverage was well over 20% of all hooks set so the reliability of this assessment is ‘high’. Distributional data suggest the majority (95%) of shy-type albatrosses killed will be juvenile and adult white-capped albatrosses (Table 1). Prior to 2002, when effort was three times greater than in 2005, the impact of the fishery would have been classified as ‘very high’ (Fig. 2(a), Ryan et al. 2002) for shy-type albatrosses.

South African Offshore Demersal Trawl Fishery

This fishery primarily targets hake (*Merluccius* spp.) and is the most valuable South African fishery. The number of participants in this fishery has increased from seven in 1986 to 61 vessels in 2001. Most effort occurs in the south and west of South Africa’s EEZ. Prior to 1999 the number of trawl hours annually was approximately

150 000 but more recently average total effort has dropped below 90 000 trawl-hours (Fig.2 (b)).

In 2004, there were 40 000 trawls (~68 800 trawl-hours) of which 121 (0.3%) were observed. During these trawls 43 birds were recorded killed by collisions with trawl warps or by drowning (Petersen et al. 2004). Of these 35% were shy-type albatrosses. This extrapolates to approximately 14 300 birds killed each year, of which 5 000 were shy-type albatrosses (Petersen et al. 2004), most of which will be white-capped albatrosses (Table 1). I assess this to be a 'very high impact' fishery for seabirds overall, and for shy-type albatrosses. However, as observer coverage was less than 10%, the reliability of this assessment is 'low'. A more comprehensive bycatch study was conducted in 2005 but the data have yet to be released.

Table 1. Seabird bycatch assessments for fisheries within the range of shy and white-capped albatrosses.

Geographic Area	Fishery	Does the fishing within the Area overlap with known distribution?								Impact assessment for fishery			Estimated potential annual mortality by species	
		Shy Albatross						White-capped Albatross						
		Albatross Island		Mewstone		Pedra Branca		All populations		Reliability	All seabirds (killed annually)	Shy-type albatrosses (killed annually)	Shy albatrosses (killed annually)	White-capped albatrosses (killed annually)
		Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults					
Area 1	South African PLF									high	very high (2 000)	high (500-600)	25-30	475-570
	South African DTF	rarely	no	yes	no	unknown	no	yes	yes	low	very high (14 300)	very high (5 000)	250	4 750
	Namibian PLF									low	high (550)	medium (190)	10	180
	Namibian DTF									low	very high (2 750)	high (960)	50	910
Area 2	Asian DWLF	rarely	no	yes	no	unknown	no	yes	yes	see below	see below	see below	see below	see below
Area 3	Australian WTBF	yes	yes	yes	yes	yes	unknown	yes	yes	low	low (20)	low (0)	0	0
Area 4	Australian ETBF	yes	yes	yes	yes	yes	yes	yes	yes	medium	high (950)	low (10)	3	7
	Australian SESSF									high	low (60)	low (12)	4	8
Area 5	New Zealand tuna (domestic)									medium	medium (300-450)	low (40-60)	0	40-60
	New Zealand tuna (foreign)									high	low (40-80)	low (1-17)	0	1-17
	New Zealand ling	no	no	rarely	no	no	no	yes	yes	medium	high/very high (500-1 500)	low (1)	0	1
	New Zealand hoki									medium	medium (270-335)	low (32-93)	0	32-93
	New Zealand squid									high	high (700-850)	high (450)	0	450
Areas 1 to 5	Asian DWLF	rarely	no	yes	no	unknown	no	yes	yes	low	v. high (17 000)	v. high (1 300)	45	1 255
Totals											39 440-40 845	8 496-8 693	387-392	8 109-8 301

Estimates of annual mortality for all seabirds, shy-type albatrosses, and shy and white-capped albatrosses separately. Species-specific estimates assume that the proportions of shy albatrosses among the shy-type bycatch in Areas 1-2 and Areas 3-4 are 5% and 34% respectively (see Methods). Bycatch of shy albatrosses in Area 5 is likely to be extremely rare. It is estimated the Asian DWLF catches approximately 860 shy and white-capped albatrosses annually in Areas 1 and 2 (see Fishery Assessment) but will predominantly impact only white-capped albatrosses in Areas 3 - 5. Zero bycatch estimates do not imply shy or white-capped albatrosses will never be caught in these fisheries.

South African PLF - South African pelagic longline fishery; South African DTF - South African demersal trawl fishery; Namibian PLF - Namibian pelagic longline fishery; Namibian DTF - Namibian demersal trawl fishery; Australian WTBF - Australian western tuna and billfish fishery; Australian ETBF - Australian eastern tuna and billfish fishery; Australian SESSF - Australian south east scalefish and shark fishery; New Zealand tuna (domestic) - New Zealand pelagic tuna fishery (domestic vessels); New Zealand tuna (foreign) - New Zealand pelagic tuna fishery (foreign vessels); New Zealand ling - New Zealand demersal ling longline fishery; New Zealand hoki - New Zealand hoki trawl fishery; New Zealand squid - New Zealand squid trawl fishery; Asian DWLF - Asian (Japanese, Korean and Taiwanese) distant water longline fisheries.

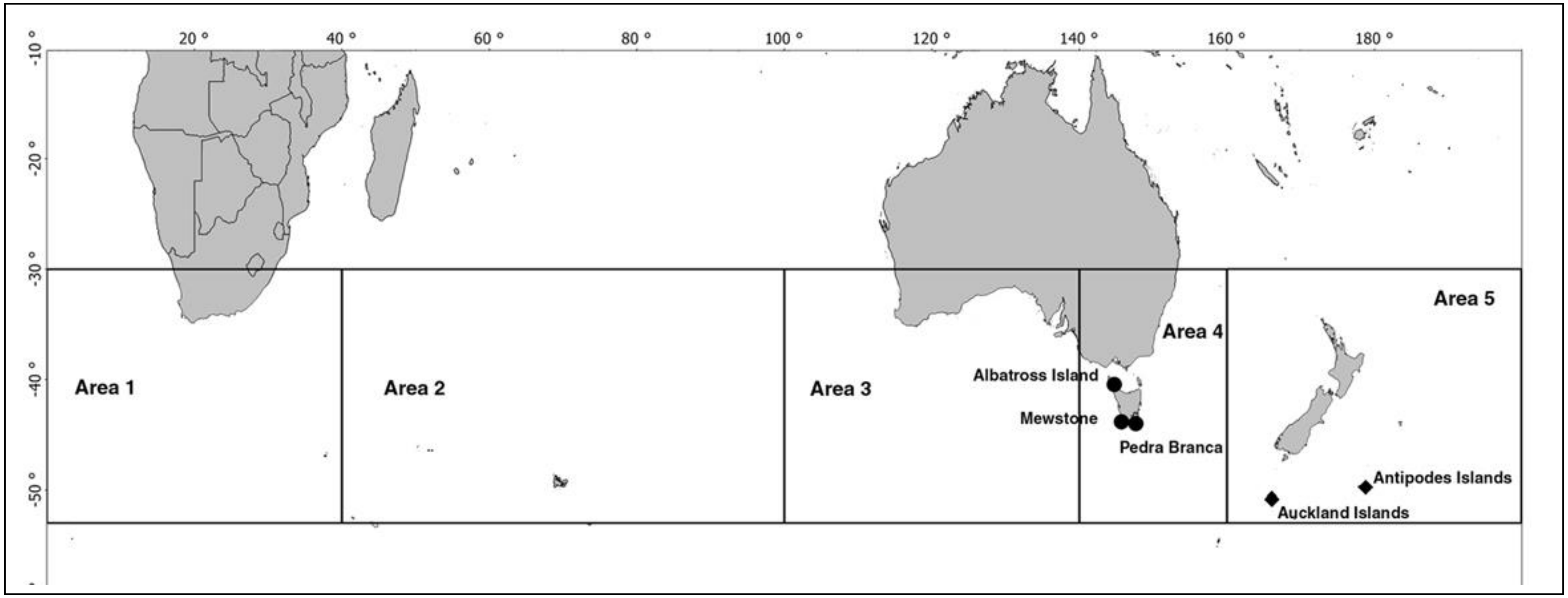

Figure 1 – The assigned areas used in the assessment of the impact of incidental mortality in fisheries, corresponding to the known and assumed foraging distribution of shy and white-capped albatrosses. Breeding sites for shy and white-capped albatrosses are indicated by circles and diamonds, respectively.

Namibian Pelagic Longline and Offshore Demersal Trawl Fisheries

A foreign longline fishery started in Namibia in 1993 targeting bigeye tuna, followed by an exploratory longline swordfish fishery in 1996. By 2003, 20 longline vessels were active in the fishery targeting bigeye tuna, swordfish and large pelagic sharks. The number of fishing days per year in 2002 and 2003 were 1 460 (~2.8 million hooks) and 1 798 (~3.4 million hooks) respectively (Fig.3 (a)). Bycatch information is very poor but observers have reported that approximately one bird is killed per 10 day trip and most are shy-type, black-browed (*Thalassarche melanophrys*) and Atlantic yellow-nosed albatrosses (*T. chlororhynchos*) (Voges, 2005).

In 1990, Namibia issued 38 licences to foreign, national and joint-venture trawl vessels to target hake within Namibia's EEZ. By 2002 this number had increased to 100 active vessels that completed 13 353 trawl-days (Fig.3 (b)). Effort data in the form of trawl-hours per year were unavailable for this fishery so to estimate bycatch I conservatively assume only one trawl-hour per fishing day (see below).

In these longline and trawl fisheries, methods are similar to those in South Africa. If bycatch rates (0.2 birds per 1 000 hooks; 0.21 birds per trawl-hour) and composition (35% shy-type albatrosses) are also similar I estimate, based on 2002 effort, approximately 550 birds (longline) and 2750 birds (trawl) are killed annually. Of these birds, 190 and 960 shy-type albatrosses are killed annually in these fisheries and most (~95%) are likely to be white-capped albatrosses. For seabirds, these are therefore 'high' (longline) and 'very high' (trawl) impact fisheries. For shy-type albatrosses these fisheries are 'medium' and 'high' impact fisheries respectively but due to virtually no observer data our confidence in these assessments is 'low'.

Major distant water pelagic longline fleets of Japan, Taiwan and Korea

This fishery was described in detail by Tuck et al. (2003). The fleets of these nations number in the hundreds of vessels and target tuna and swordfish species on the high seas. Between 1995 and 2000, effort has exceeded 100 million hooks annually across all five areas (Fig.4) but effort varies greatly by area and season. Within Areas 1, 3, and 4 effort was relatively uniform between 1995 and 2000 with 41.5, 19.4 and 12.1 million hooks set respectively. In Areas 2 and 5 effort fluctuated but averaged 35.5 (Area 2) and 7 million hooks set (Area 5). In Areas 1, 2, 4 and 5, most (78 – 96%) hooks were set during the austral winter (April-September; Fig.4) whereas in Area 3 most (80%) hooks were set during the third and fourth quarters (Fig.4 (c)).

Between 1995 and 2003 the Japanese fleet was reported to have killed between 6 000 and 9 000 seabirds each year (Government of Japan, 2004; Kiyota and Takeuchi, 2004). A further analysis of the same data set indicated that in 2001 and 2002 when approximately 3% of hauls were observed, the estimated annual total seabird bycatch was 6 516 (95% CI 3 376–10 378; 2001 data) birds (0.14 birds/thousand hooks) and 6 869 (95% CI 3 811–10 213; 2002 data) birds (0.18 birds/thousand hooks) (CCAMLR, 2005). In 2001 and 2002, 74.1% of the seabirds reported killed were albatross species, and of those albatrosses identified to species (n = 281), 28 (10%) were reported to be shy-type albatrosses (Kiyota and Takeuchi, 2004; CCAMLR, 2005). Seabird bycatch information for both the Korean and Taiwanese pelagic longline fleets is poor, but these fleets employ similar vessels and gear to the Japanese, so a similar bycatch rate can be reasonably assumed. Based on mean effort between 1995 and 2000, these data suggest that a total of approximately 17 000 seabirds are killed each year. Of these birds, 460, 400, 220, 140 and 80 shy-type albatrosses are killed annually in Areas 1 to 5 respectively. Collectively this is a ‘very high impact’ fishery but the reliability of this assessment is ‘low’ due to poor observer coverage and extrapolation of bycatch rates between vessels of different origins. This fishery primarily overlaps with the distribution of juvenile and adult white-capped albatrosses, and also juvenile shy albatrosses (Table 1). In Areas 3 and 4 shy albatrosses from all colonies forage mainly within Australia’s EEZ (Hedd et al. 2001) and would therefore have minimal contact with this fishery.

Australian pelagic tuna fisheries

The Eastern Tuna and Billfish Fishery (ETBF; Area 4) and Western Tuna and Billfish Fishery (WTBF; Area 3) commenced operations in the 1980s but effort remained at low levels until 1997, when they expanded rapidly following the exclusion of the Japanese tuna fishery from the EEZ (Environment Australia, 1998). Target species are yellowfin tuna, big eye tuna, albacore, southern bluefin tuna (*T. maccoyii*), and broadbill swordfish. Although there are a large number of licence holders in both fisheries (147 in ETBF, 90 in WTBF), the number of active vessels in 2005 was significantly fewer (90 in ETBF; 4 in WTBF).

Since 1998, effort in the ETBF has remained relatively constant at around 3.4 million hooks (Fig.5 (a)). Most effort occurred between latitudes 30 to 35^{o}S and in the first two quarters of each year. In the WTBF effort has declined from a peak of 4.2 million hooks in 1999 to 1.0 million hooks in 2004 (Fig.5 (b)) following poor market conditions and low fish abundance. Most effort (55%) occurs in the first two quarters of each year.

Between 2001 and 2004, 1.2 million hooks (11.8 % of hooks set) in the ETBF were observed and indicated a bycatch rate of 0.28 birds per 1 000 hooks (Baker and Wise 2005). Of 346 birds observed caught and retained, only 3 (0. 9%) were shy-type albatrosses (Barry Baker and Rosemary Gales, unpublished data). Based on current (2004) effort these data suggest approximately 10 shy-type albatrosses will be killed in this fishery each year. Within the WTBF the bycatch of seabirds during 2002–2004 was 0.02 birds per 1 000 hooks, but only 200 000 hooks (4% of effort) were observed. No albatrosses were observed caught during this period (Australian Fisheries Management Authority, unpublished data). This low bycatch has been attributed to the four active vessels all fishing at night to target broadbill swordfish.

These data suggest that almost 1 000 birds may be caught in these fisheries annually, of which very few are shy-type albatrosses. The ETBF is classed as a 'high' impact fishery for seabirds, and at the current level of effort, both ETBF and WTBF are likely ’low’ impact fisheries for shy-type albatrosses. I assign ‘medium’ and ‘low’ reliability to the assessments for the ETBF and WTBF respectively. These fisheries overlap with the distribution of juvenile and adult shy and white-capped albatrosses (Table 1).

Australian Southern and Eastern Scalefish and Shark Fishery (SESSF), (Scalefish Hook Sector)

This is a multi-sectored fishery that uses demersal gillnets, drop lines, demersal longlines and traps. The Scalefish Hook Sector of this fishery is concentrated in continental slope waters around Tasmania and eastern Victoria and targets demersal finfish such as pink ling (*Genypterus blacodes*) and blue eye trevalla (*Hyperoglyphe antarctica*). This Sector is the only one considered likely to impact shy-type albatrosses. Effort has increased rapidly from 670 000 hooks in 2000 to 7.15 million hooks set in 2004 with 70% of hooks set during the austral summer (Fig.5 (c)). In 2005 there were seven auto-longliners vessels active in the fishery each limited to deploying 15 000 hooks per day.

Since 2002 vessels have been required to have a fisheries observer on board for every fourth trip. Between 2002 and 2005, 2.01 million hooks were observed and bycatch was 0.01 birds per 1 000 hooks. Three of the 16 birds observed caught were shy-type albatrosses (Australian Fisheries Management Authority, unpublished data). All birds were caught by one vessel, which subsequently adopted integrated weight longlines which can dramatically reduce seabird bycatch (Robertson et al. 2006). I assess this to be a ‘low’ impact fishery for both seabirds and shy-type albatrosses (high reliability assessment). This fishery overlaps with all life stages of shy and white-capped albatrosses (Table 1).

New Zealand Pelagic Tuna Fishery (domestic and foreign charter vessels)

This fishery operates mainly off the west coast of the South Island and along the east coast of the North Island (Area 5). In 2002 there were 156 vessels in the fishery, 67% of which were longliners (Government of New Zealand, 2006). The fishery targets tuna species and has expanded rapidly from 3.5 million hooks in 1998 to 10 million hooks in 2002 (Fig. 5 (a)). Most effort (58%) occurs in the first two quarters of the year.

Domestic effort data were not available for 2003 and 2004 but observer effort increased to 8% and 14% (Government of New Zealand, 2006) when Waugh and MacKenzie (2006) estimated total annual bycatch to be 439 (95 % CI 110 – 1 293; 2003 data) and 322 seabirds (95 % CI 124 – 799; 2004 data). Waugh and MacKenzie (2006) report the overall bycatch composition for domestic and foreign tuna fisheries and state that 13% (17/133) of the seabirds caught by observed vessels in 2003 and 2004 were shy-type albatrosses. Consequently I estimate that between 40 and 60 shy-type albatrosses are killed in this fishery annually. Waugh and Mackenzie (2006) note, however, that there was very little coverage of domestic sets off the east coast of New Zealand's North Island where most of the domestic effort occurs. Domestic effort in this fishery therefore has a 'medium' impact assessment for seabirds, but a 'low' impact assessment for shy-type, in this case, white capped albatrosses.

In addition to New Zealand domestic effort, between 1998 and 2002, Japanese charter vessels set between 0.6 and 1.1 million hooks each year with most effort (92%) confined to April-June (Fig.6 (b)). A maximum of four vessels operated in the fishery, targeting mainly southern bluefin tuna (New Zealand Ministry of Fisheries, 2004). Baird (2004; 2005), based on high levels of observer coverage (>90% of hooks set), estimated the total number of seabirds killed by Japanese charter vessels in 2001/02 and 2002/03 was 81 (coefficient of variation (CV) 4%) and 42 (CV 6%) respectively. Of these, it is estimated 17 (21%) and 1 (2%) were shy-type albatrosses respectively (Baird 2004, 2005), all of which are likely to be white-capped albatrosses (Table 1). These data suggest this is a 'low' impact fishery and I assign 'high' reliability to this assessment.

New Zealand Demersal Ling Longline Fishery

In 2001/02 and 2002/03 (by NZ fishery year) over 40 vessels reported targeting ling (*Genypterus blacodes*) but most (c.90%) effort can be attributed to six vessels that used auto-longline gear (Baird 2004, 2005). Between 1998 and 2002 an average of 28.9 million hooks were set annually and most (61%) were set between July and December (Fig.6 (c)). Effort is usually concentrated east of New Zealand towards the Chatham Rise and Bounty Platform (Baird 2004, 2005).

In 2001/02 and 2002/03 total bycatch was estimated to be 1 450 (CV 16%) and 543 (CV 10%) birds, respectively (Baird 2004, 2005), resulting in a seabird impact assessment of ‘high’ to ‘very high’. While most birds were white-chinned (*Procellaria aequinoctialis*) and grey petrels (*P. cinerea*), at least 20% of the bycatch comprised albatrosses (Robertson et al. 2003a). However, of over 520 birds retained for identification since 1998, only one has been confirmed as a shy-type albatross (Robertson et al. 2003a). Effort in this fishery only affects white-capped albatrosses and at present the overlap is minimal. I assess this to be a ‘low’ impact fishery with ‘medium’ reliability (15% observer coverage in 2001/02). However, any spatial shift in effort towards the west and south of the South Island of New Zealand will substantially increase the impact on white-capped albatrosses

New Zealand Hoki Trawl Fishery

Hoki (*Macruronus novaezelandiae*) is New Zealand's most abundant commercial fish species and is targeted by a major offshore trawl fleet including small, domestic and larger, foreign-owned trawlers. In 2002/03 (by NZ fishing year), 67 vessels operated in the fishery (Baird, 2005), 39 of which were domestic vessels that accounted for 75% of all effort. About 70% of all tows used bottom trawl nets, with the remainder deploying mid-water nets (Baird 2005). Bottom nets were responsible for 63% of all observed seabirds caught in 2002/03 (Baird 2005). Hoki is fished all year round, but most fishing effort (58%) is concentrated in a ten week period between June and September. Between 1998 and 2002 annual effort remained constant at 105 000 to 125 000 trawl hours each year (Fig.6 (d)).

In 2002/03 10% of all tows were observed, with the coverage extending to 48% of the 67 vessels that operated in the fishery (Baird 2004, 2005). Baird (2004, 2005) considered that data were only sufficient to estimate total annual seabird bycatch in the main fishing areas and estimated that 334 (CV 33%) and 269 (CV 23%) birds were killed in 2001/02 and 2002/03 respectively. This fishery therefore is assessed as having a 'medium' seabird impact. Of the 29 and 42 seabirds observed killed and subsequently identified for these years, 8 (28%) and 5 (12%) respectively were shy-type albatrosses all of which are likely to be white-capped albatrosses (Table 1). These data suggest fewer than 100 white-capped albatrosses were killed in the 2002/03 fishery-year and this is therefore a 'low' impact fishery (medium reliability) for these birds.

New Zealand Squid Trawl Fishery

This fishery mainly operates off the South Island of New Zealand on the Stewart-Snares Shelf and the Auckland Islands and through bottom-trawling mainly targets arrow squid (*Nototodarus sloanii*). This fishery uses large foreign-owned (Japanese, Korean, Ukrainian and Polish) vessels under charter to New Zealand fishing firms and has caught between 30 000 and 60 000 tonnes of squid in most years since 1986. Most (86%) effort occurs between January and March and exceeded 4 000 trawl hours in 1998, 2000 and 2002 (Fig.6 (e)). Not all vessels use meal plants or retain offal and hence squid trawlers are regularly attended by large numbers of seabirds.

Approximately 22% of tows by foreign vessels were observed in 2002 (Baird 2004). Data were insufficient to estimate bycatch across the entire fishery but Baird (2004; 2005) estimated the total number of seabirds killed in 2001/02 and 2002/03 in main fishery areas was 710 (CV 11%) and 841 (CV 12%) respectively. This fishery is therefore ranked as having a 'high' impact on seabirds. Of the 110 and 185 seabirds observed killed and subsequently identified for these years, 60 (55%) and 110 (59%) respectively were shy-type albatrosses. Frequent interactions with trawl warps occur and most of the recorded shy-type albatross bycatch in New Zealand occurs in this fishery (Robertson et al. 2003a). These data suggest approximately 450 shy-type albatrosses were killed annually within the main fishery areas during the 2001/02 and 2002/03 fishery-years, all of which are likely to be white-capped albatrosses (Table 1). However, mortalities were only based upon bycatch in 'main fishery areas' and so total bycatch is likely to be higher than stated here. For shy-type albatrosses, I therefore assess this to be a 'high' impact fishery. Over 20% of tows were observed in 2002 so the reliability of this assessment is 'high' although observer data may be biased geographically.

Discussion

Lewison et al. (2005) identified the need to consider the effects of bycatch on globally distributed species across large ocean regions. For each fishery, I have assigned the level of impact to seabirds in general and then, by focussing on shy and white-capped albatrosses only, produced species-specific impact statements. This is the first time such assessments have been made for any seabird species. These evaluations confirm drastically different degrees of vulnerability to bycatch mortality between shy and white-capped albatrosses (Table 1).

In the last decade, longline fishing has been identified as the most pervasive threat to seabirds, causing widespread declines in populations (Brothers 1991, Robertson and Gales 1998, Nel et al. 2002). More recently, the mortality of seabirds caused by trawlers has been increasingly recognised as another significant cause of population declines (Weimerskirch et al. 2000; Sullivan and Reid 2003; Sullivan et al. 2006). Trawl fishery bycatch was initially characterised by collisions with netsonde cables, although seabirds are also killed by collisions with warps and entanglements in nets. Of the fisheries assessed here, total annual seabird bycatch levels were similar between fishing methods (trawl 45%, longline 55%). However, for shy-type albatrosses, the majority of birds (75%) were killed in trawl fisheries, largely reflecting the very high levels of albatross mortality inflicted by the South African and Namibian trawl fisheries.

More specifically, of the 13 fisheries assessed, six were ranked as having a 'very high' impact assessment for seabirds generally, each killing an estimated 1 000 birds or more every year. Two of these, the South African Demersal Trawl Fishery and the Asian distant-water longline fishery, were each estimated to kill > 10 000 seabirds each year.

For shy-type albatrosses, five fisheries were identified as 'high' or 'very high' impact, collectively killing over 8 500 shy-type albatrosses annually. Of these, the highest impact fishery (60% of total estimated mortality) was the South African Demersal Trawl Fishery; followed by the Asian distant-water longline fishery (15%), the Namibian Demersal Trawl Fishery (11%), the South African Pelagic Longline Fishery (7%) and the New Zealand Squid Trawl Fishery (5%). However, only four of the thirteen fisheries achieved a 'high' reliability assessment. It is therefore clear that current understanding of the true level of global bycatch remains inadequate, and my estimates could significantly under- or over-estimate the true number of shy-type albatrosses killed.

Impact of current fishing effort on shy albatrosses

The range of adult shy albatrosses is largely confined to waters around Tasmania and southern Australia (Brothers et al. 1997; Brothers et al. 1998; Hedd et al. 2001). This is where most of an estimated 900 shy-type albatrosses were killed by Japanese tuna longliners between 1988 and 1997 (Gales 1998), the majority (~ 65%) being adult shy albatrosses (Abbott et al. 2006). Corresponding annual effort from both Japanese and Australian fishing vessels in Australian waters (Area 4) during this period exceeded 10 million hooks, and probably led to a level of mortality of shy albatrosses that would have been unsustainable in the long term. Since then however, pelagic tuna longlining effort in southern Australian waters has changed dramatically; Japanese effort ceased in 1997 and current domestic effort in eastern Australia is concentrated in northern waters where the likelihood of encountering albatrosses is much lower (DPIWE, 2004). As a result the bycatch of albatrosses has been reduced to low levels (<0.02 birds per thousand hooks). Only one other fishery, the Scalefish Hook Sector of the Southern and Eastern Scalefish and Shark Fishery, has been observed to kill more than one shy albatross in Area 4 since 1998, and this led to operational changes by the single vessel involved to avert further captures. I conclude that currently the distribution of all adult shy albatross only overlaps with low impact fisheries, and hence adults are not greatly impacted by fisheries-related mortality.

Juvenile shy albatrosses, from the Mewstone colony in particular (Brothers et al. 1997), are known to have a much wider range than adults. The proportion of juveniles that migrate to South African waters from each of the colonies is not known but banding and satellite tracking data suggest that birds from the Mewstone colony regularly traverse the Indian Ocean. This behaviour will bring them into contact with four 'high' or 'very high' impact fisheries in Areas 1, 2 and 3, where most of the estimated total annual bycatch of < 400 shy albatrosses occurs (Table 1). The population status and trend for the Mewstone colony remains unknown, although the Albatross Island population has undergone significant recovery since the decimation of the colony by 19th century sealers (Gales 1998; Rosemary Gales unpublished data).

Impact of current fishing effort on white-capped albatrosses

Currently, the vulnerability of both adult and juvenile white-capped albatrosses to fisheries bycatch mortality appears to be much greater than for shy albatrosses, as they occur over a wider geographic range throughout their lives. Abbott et al. (2006) found that among bycatch samples white-capped albatrosses greatly outnumber shy albatrosses in all areas outside of Area 4, reflecting their foraging distribution and consistent with their greater global population size (Gales 1998; Ryan et al. 2002; Robertson et al. 2003b). White-capped albatrosses are exposed to the four ‘high’ or ‘very high’ impact fisheries that operate on the high seas and around southern Africa. Analysis of bycatch specimens has shown that most (>95%) of the shy-type albatrosses killed in South African fisheries are white-capped albatrosses(Abbott et al. 2006; MCD and PGR, unpublished data). Even when breeding, white-capped albatrosses overlap with the squid trawl fishery operating south of New Zealand, and all analysed shy-type bycatch specimens from this fishery were white-capped albatrosses (Abbott et al. 2006). Bycatch specimens from the distant-water Asian longline fleets were not available for examination by Abbott et al. (2006) but the large number of hooks set across the range of the white-capped albatross in fisheries using minimal mitigation measures are grounds for serious concern. I conclude that most shy-type albatrosses killed annually are white-capped albatrosses (estimated annual mortality 8 109 —8 301 birds, Table 1). At this level the current risk to white-capped albatrosses is high, and may be unsustainable for some or all of the smaller colonies.

Conservation and management implications

The clear difference in the current level of risk imposed by fisheries bycatch on shy and white-capped albatrosses poses a conservation conundrum. While adult and, to some extent, juvenile shy albatross are currently not at high risk from fisheries interactions, the species is well studied, whereas the opposite is the case for the white-capped albatross. Long-term demographic data have been collected on shy albatrosses for over 25 years (Department of Primary Industries and Water, Tasmania unpublished data). Information is also available on their foraging behaviour (Hedd et al. 1997; Hedd et al. 2001), at-sea distribution (Brothers et al. 1997; Brothers et al. 1998; Hedd et al. 2001), breeding biology (Gales 1993) and genetic structure (Abbott and Double 2003a, b). In contrast, the white-capped albatross is significantly impacted by fisheries interactions across much of its wide range, with all age classes at risk. However, little is known about their population status, breeding biology, life history and at-sea distribution (Robertson et al. 2003b). As there are no accurate estimates of population size for this species, there can be no reliable assessments of status or trends (Gales, 1998). Much of this information is critical for informing management on how bycatch levels are affecting population dynamics. In the absence of information on population size and status and numbers of birds killed, it is difficult to confidently assess if the current bycatch level is sustainable.

Data on population size and status of little studied white-capped albatrosses are urgently needed for all colonies. Knowledge of temporal and spatial variation in their at-sea distribution is required to assess the temporal and spatial impact of fisheries-related mortality. Recent initiatives from the New Zealand Ministry of Fisheries and Department of Conservation to commence such studies should go some way to address to redress these data deficiencies.

To understand the conservation implications of fisheries bycatch mortality on shy and white-capped albatrosses, improved knowledge of levels of bycatch in all major fisheries known to kill shy-type albatrosses is also needed. The independent observer coverage in most of the fisheries examined here is either non-existent or falls below the level required to accurately estimate bycatch levels (Agnew, 2001). It is also important that fisheries observers retain all seabirds killed in fishing operations and return carcases for analysis to determine species, age, sex, breeding status and, where possible, provenance. This is a mandatory requirement in CCAMLR and some Australian and New Zealand longline fisheries, and is essential in assessing risk to species and improving knowledge of fishery impacts (Environment Australia 1998; Gales 1998; Abbott et al. 2006).

Observer programs and bycatch risk assessments need to be maintained and regularly reviewed. Spatio-temporal effort in fisheries is dynamic and fluctuates in response to market forces and the status of target stocks. Changes in effort can rapidly change the impact upon bycatch species. For example, while the risk to shy albatrosses is currently low, it would require only slight changes in the distribution of fishing effort and practices for this situation to change. Implementation of observer programs is also essential because quantifiable evidence forces ownership of a bycatch problem onto the fishery, in many cases leading to effective management action. I know of no examples where fisheries managers have implemented mandatory mitigation measures as a precautionary measure in the absence of clear evidence of a bycatch problem.

The effective implementation of mitigation measures, and the collection and analysis of catch and bycatch data by commercial fisheries needs to be seen by industry stakeholders as an essential tool for managing impacts on both target and non-target species, and not as a financial burden to be avoided wherever possible. Despite wide acknowledgement that bycatch is the most serious threat facing many seabird species, voluntary uptake of effective seabird bycatch mitigation measures is virtually non-existent in global fisheries.

The seabird impact assessments provided here clearly illustrate the differential impact of fisheries on two different species and population cohorts. I have focussed on shy-type albatrosses but the overall seabird impact assessments show that many other seabird species are also impacted, some to levels that are not sustainable (Baird 2004, 2005; Baker and Wise, 2005; Petersen et al. 2006). For example, CCAMLR (2005) recently reviewed bycatch in the Japanese Southern Bluefin Tuna Fishery (Asian distant-water fishery) and concluded that as many as 3 000 grey-headed albatrosses (*Thalassarche chrysostoma*), 1 370 black-browed albatrosses (*T. melanophrys*), 690 giant petrels (*Macronectes spp*.) and 600 *Procellaria* petrels may be killed each year. The conclusions I have drawn about the need for urgent implementation of mitigation measures, effective evaluation of fisheries and collection of biological data to assess impacts are as critical for other seabird species as they are for shy and white-capped albatrosses.

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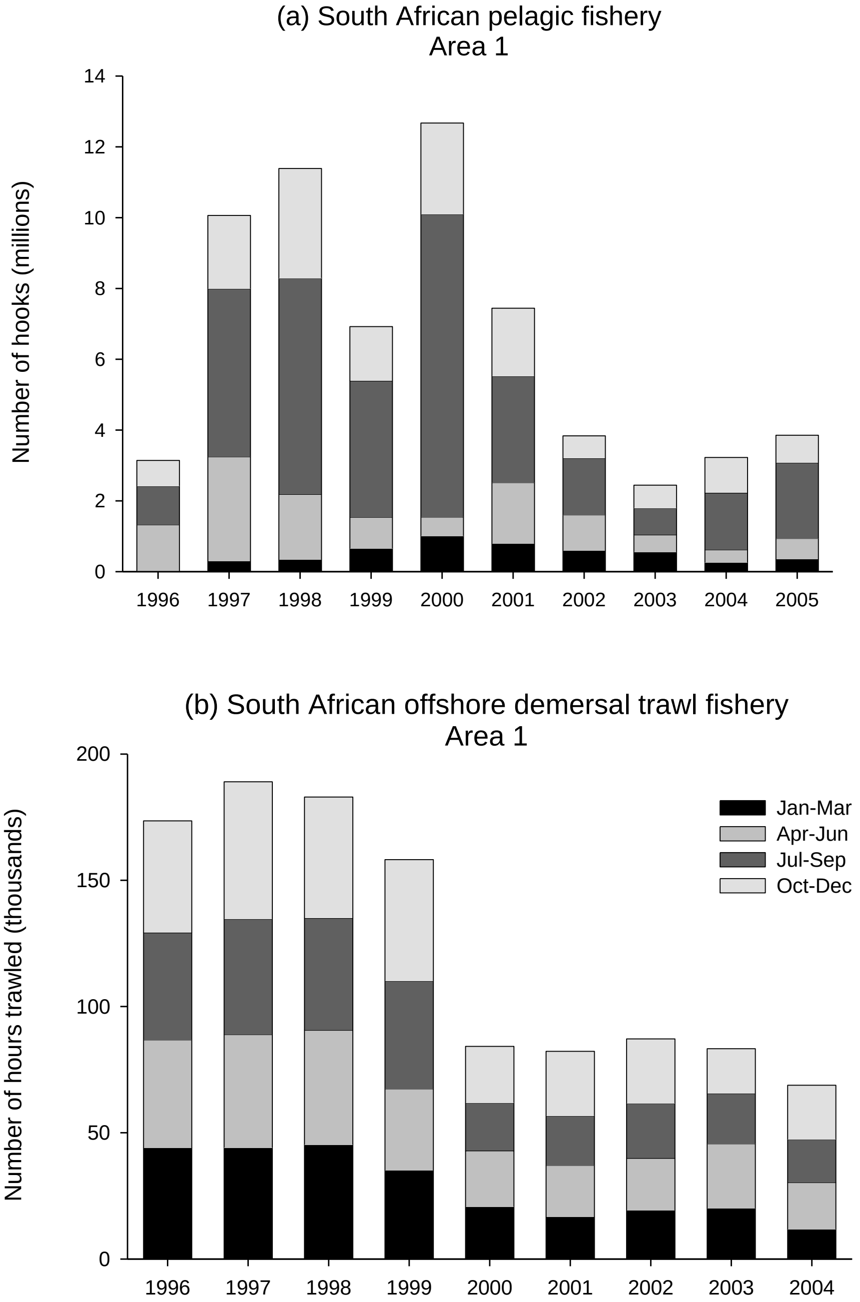

Figure 2 – (a).The total number of reported hooks set by quarter within the South African pelagic longline fishery and (b) the total number of trawling hours reported within the South African offshore demersal trawl fishery. Source: Department of Marine and Coastal Management, South Africa.

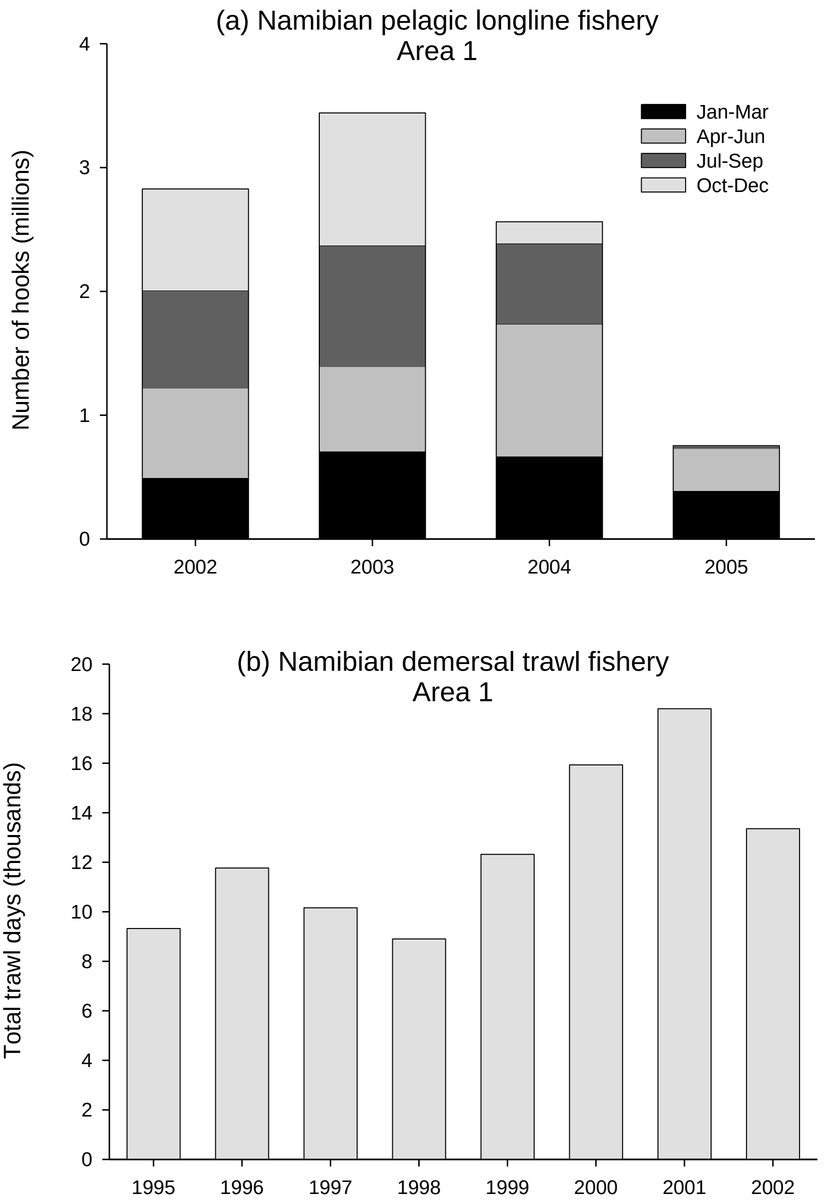

Figure 3 – (a) The total number of reported hooks set by quarter within the Namibian pelagic longline fishery and (b) the total number of trawling days reported per year within the Namibian offshore demersal trawl fishery. Source: unpublished data, Ministry of Fisheries and Marine Resources, Namibia.

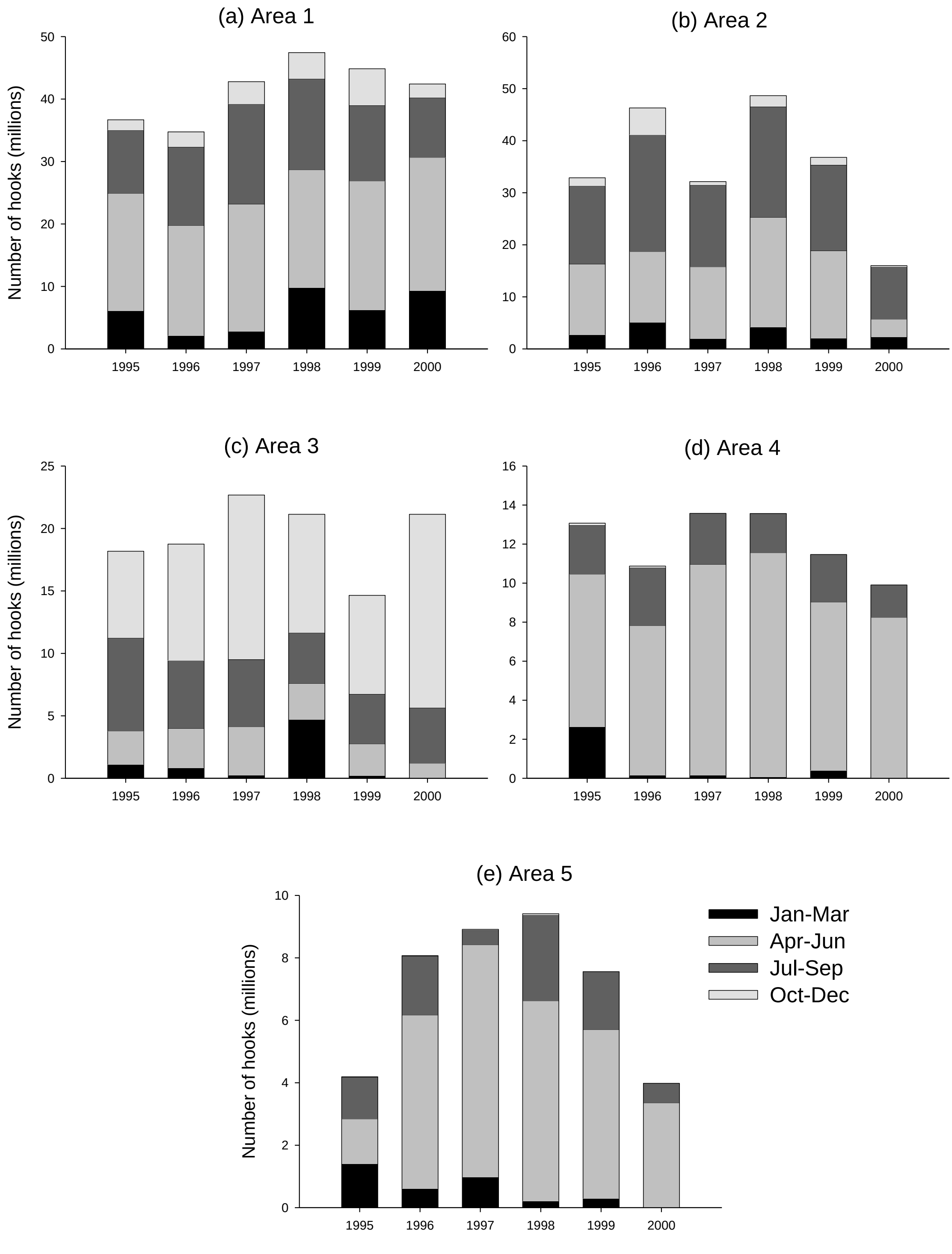

Figure 4 – The number of reported hooks set by quarter between 1995 and 2000 for the distant-water longline nations of Japan, Taiwan and Korea south of 30 S within Areas 1 to 5 (see Fig.1). Source: ICCAT, SPC, IOTC, CCSBT.

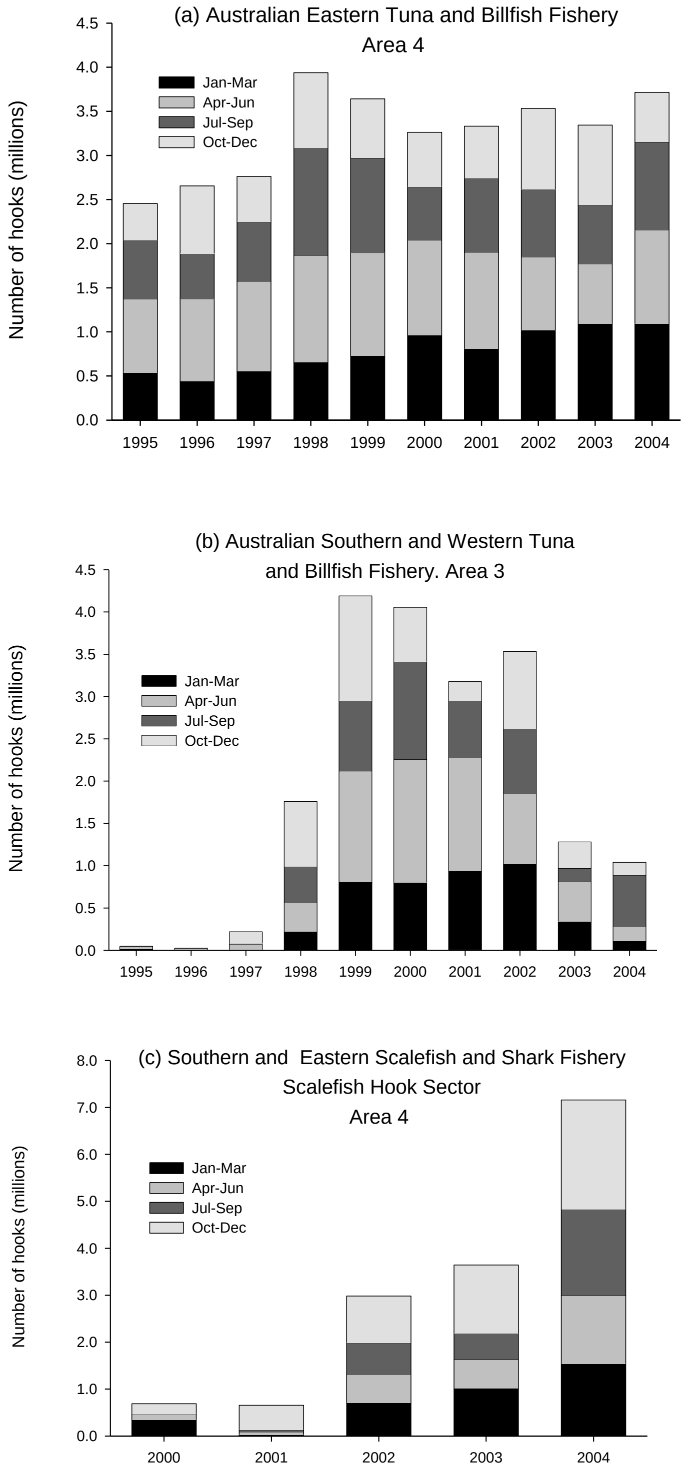

Figure 5 – The number of reported hooks set south of 30 S by quarter between 1996 and 2004 for (a) the Eastern Tuna and Billfish pelagic longline fishery of Australia, (b) the Western Tuna and Billfish pelagic longline fishery; and between 2000 and 2004 for (c) the Southern and Eastern Scalefish and Shark Fishery (Scalefish Hook Sector). Source: Australian Fisheries Management Authority.

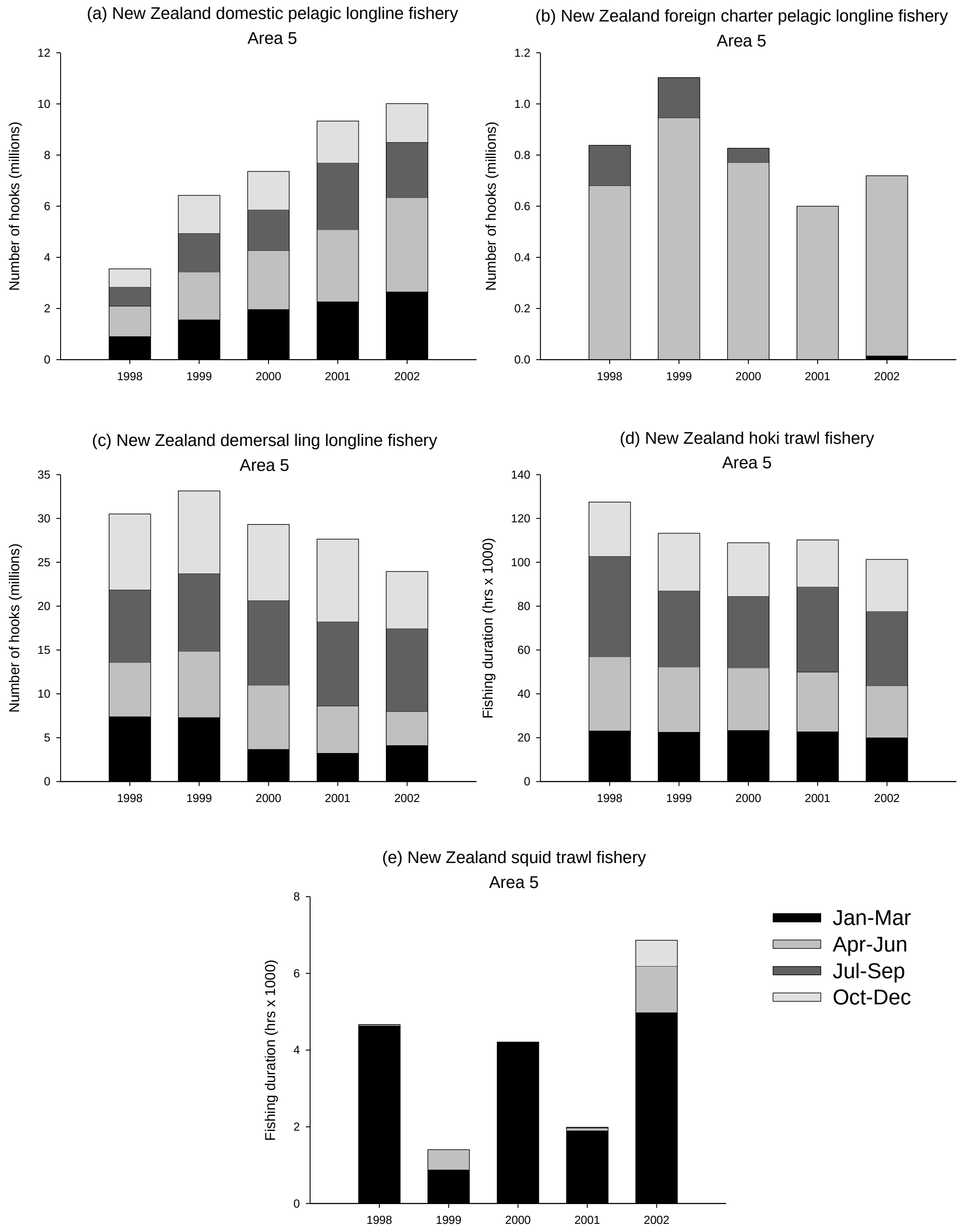

Figure 6 – The number of reported hooks set by quarter between 1998 and 2002 in New Zealand longline (a, b) and trawl fisheries (c, d and e) south of 30 S. Source: New Zealand Ministry of Fisheries

Chapter 2

Population assessment of white-capped albatross *Thalassarche steadi*

Abstract

White-capped albatrosses *Thalassarche steadi* are endemic to New Zealand, breeding mainly in the sub-Antarctic Auckland Islands. Virtually all aspects of the biology and ecology of white-capped albatrosses are poorly known and there are no well-documented population estimates available. The species is commonly caught as bycatch in global longline and trawl fisheries, and it has been estimated that 8 000 birds are killed each year, a level that may well be unsustainable. From 2006 to 2013 I undertook annual counts of breeding pairs of white-capped albatrosses on the Auckland Islands using aerial photography. These censuses were carried out in either December or January to estimate population size and track population trends. Counts from a series of 'close-up' photographs taken each year indicated that the number of loafing birds present in the colonies differed between the early and late incubation periods of the breeding season. The proportion of loafers was very low in early incubation (1-2% of birds present), but higher later in the incubation period (14% of birds present). Estimated annual counts for all three breeding sites in the Auckland Islands were adjusted to account for the presence of loafers, giving adjusted estimates of annual breeding pairs for each year from 2006 to 2013. The mean number of annual breeding pairs in the Auckland Islands during this period was 90 141, with annual estimates ranging from 73 838 to 116 025. Trend analysis for all sites combined using regression splines showed no clear evidence for monotonic decline over the eight years of the study, providing insufficient evidence to reject the null hypothesis of no trend in the total population. Trend analysis using Program TRIM, currently used by the Agreement on the Conservation of Albatrosses and Petrels to assess population trends in albatross populations, indicated an average growth rate of -3.16% per year, assessed by TRIM as moderate decline. However, a simple linear trend analysis as performed by TRIM is not well suited to a data set with high inter-annual variability. Continuation of annual monitoring is recommended to clarify the population's status and determine if current high levels of fishing mortality are sustainable.

Introduction

Accurate estimation of numbers and the establishment of monitoring programmes is critical for determining conservation status, and for identifying the key factors influencing changes in population size and demography of seabirds and other vertebrates. Diagnosing the causes of declines or increases is much harder without information on the timing and magnitude of population changes observed (Wolfaardt and Phillips 2012).

Methods described in the literature for counting birds and mammals are many and various (e.g. Caughley 1980; Bibby et al. 1992; Caughley and Sinclair 1994; McCallum 2000). They include direct ground or aerial counting of visible animals (Bibby et al. 1992), aerial survey using light aircraft or drones as a viewing or photographic platform (Arata et al. 2003; Robertson et al. 2007; Sarda-Paloma et al. 2011), eliciting responses through the use of playback of recorded calls (Conway et al. 2004), use of dogs (Bibby et al. 1992), capture-mark-recapture techniques using bands or radio-tags (White and Garrott 1990), territory marking and intensive ground searches (Bibby et al. 1992), ground counts of nests and burrows to infer presence (Congdon et al. 2007), use of remotely operating cameras (Southwell et al. 2010) and a myriad of other approaches.

Wolfaardt and Phillips (2012) recently summarised methodologies relevant to surface-nesting albatrosses and petrels and noted that, with recent technological advances in photographic equipment and image processing software, aerial photography has become much easier to use and represents a more accurate method of surveying breeding populations. It is becoming increasingly preferred as the census method of choice for surface nesting seabirds, especially in remote locations where access is difficult because of logistical or topographical constraints (Wolfaardt and Phillips 2012).

White-capped albatrosses *Thalassarche steadi* are endemic to New Zealand, breeding on Disappointment Island, Adams Island and Auckland Island in the

Auckland Island group (Figure 1), Bollons Island in the Antipodes Island group (Gales 1998), and occasionally at the Forty-Fours in the Chatham Islands group (ACAP 2011). The population estimates of Gales (1998) suggest most (95%) of the global population breeds on Disappointment Island, an area where access is restricted to maintain environmental values at the site. Virtually all aspects of the biology and ecology of white-capped albatrosses are poorly known and although approximate population sizes are given above there are no well-documented population estimates for any of the colonies (Taylor 2000). Ground and aerial photographs have been previously undertaken of the Disappointment Island colony in 1972, 1981, 1985, 1990 and 1993 by others (Taylor 2000), but no reports or published results have been produced from these surveys.

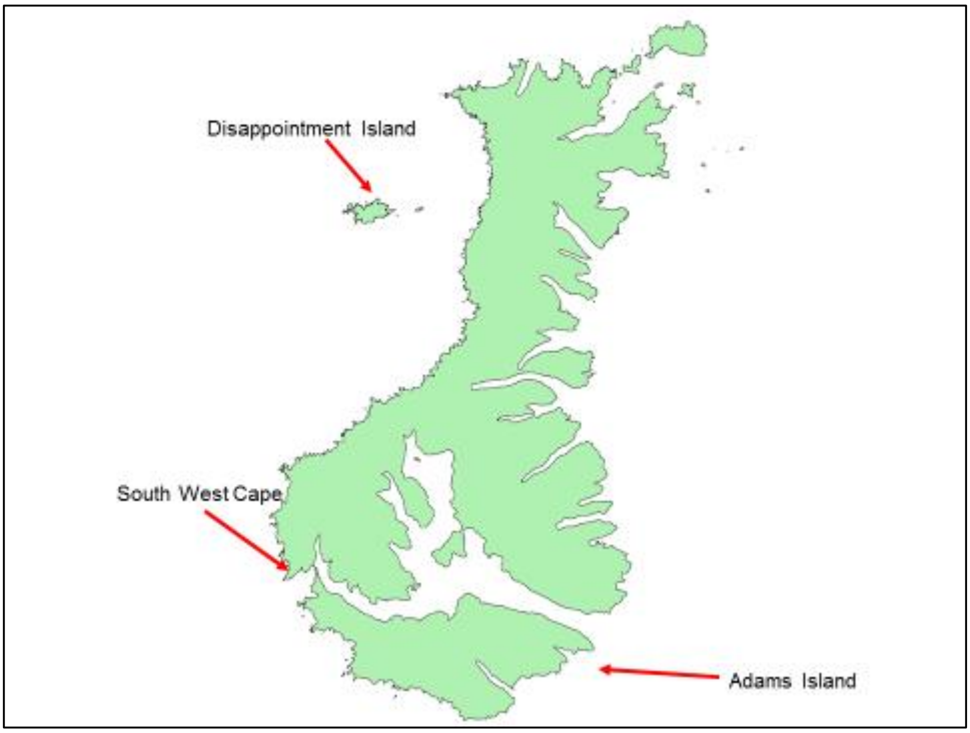

Figure 1 Breeding sites of white-capped albatross in the Auckland Islands Group. Most birds breed on Disappointment Island, with smaller colonies located on South West Cape, Auckland Island, and Adams Island.

White-capped albatrosses are known to be killed in both trawl and longline fisheries across a wide spatial scale with both adults and juveniles highly vulnerable to bycatch. (Abbott et al. 2006; Baker et al. 2007). Baker et al. (2007) estimated that 8 000 may be killed annually, although the reliability of this estimate was low due to the paucity of comprehensive observer data in most fisheries. At this level the current risk to white-capped albatrosses may be unsustainable, but, as there are no accurate estimates of population size for this species, there can be no reliable assessments of status or trends (Gales 1998). Such information is critical for informing management on how bycatch levels are affecting population dynamics. To address this deficiency, from 2006 to 2013 I undertook annual counts of white-capped albatrosses breeding in the Auckland Islands using aerial photography. These counts were carried out in either December or January to estimate the number of breeding pairs and track population trends. Here I report on the results of this study and discuss the demographic implications for the species.

Methods

Nomenclature

In the taxonomic revision of albatrosses suggested by Robertson and Nunn (1998), the shy albatross complex was split into four species: Salvin's albatross (*Thalassarche salvini*); Chatham albatross (*T. eremita*); white-capped albatross (*T. steadi*) and the shy albatross (*T. cauta*). The recognition of the latter two taxa as separate species was supported by later morphometric, phylogenetic and population genetic studies (Abbott and Double 2003a, b; Double et al. 2003). I therefore use the names 'shy albatross' and 'white-capped albatross' and refer to them as species.

Definitions

The purpose of this study was to estimate the number of pairs of white-capped albatrosses breeding in the Auckland Islands each year (annual breeding pairs). The term breeding pair is not consistently defined across studies, leading to errors in interpretation of results and detection of population trends. In this thesis I have defined it as follows:

Annual breeding pair – any pair of albatrosses that lays an egg in the breeding season of interest.

Loafers –birds present in a colony but which do not appear to be associated with an active nest at the time of observation. These birds may non-breeding birds or breeding birds away from their nest, or birds that have laid an egg earlier in the breeding season and subsequently lost it through breakage or predation.

Biology

Despite being New Zealand's most numerous breeding albatross species, very little is known of white-capped albatross breeding biology and at-sea distribution. Birds breed in the Austral spring, commencing egg-laying in mid-November, with hatching underway by mid-January, extending into early February. Chicks are guarded for approximately three weeks, and fledge in June (Thompson et al. 2011). The breeding frequency is uncertain, however Francis (2010) reported the probability that a bird that bred in one year would also breed in the next year to be 0.63, whereas the probability that a bird that didn't breed in one year but which would breed in the next year was 0.78. These results, together with observations of birds breeding in successive years, suggests that the white-capped albatross has an intermediate breeding strategy between annual and biennial (Thompson et al. 2011).

Information on the geographical range of white-capped albatross is confounded by its resemblance to the shy albatross *Thalassarche cauta* (Double et al. 2003), and there have been no published broad-scale satellite tracking or banding studies that accurately define their at-sea distribution. However, Abbott et al. (2006) used molecular species assignment methods to distinguish 'shy-type' albatross carcasses obtained from fisheries bycatch in Australia, New Zealand, and South Africa waters, thus providing some information on the geographic distributions of these species. Although information is limited, during the breeding season *T. steadi* is thought to forage mainly within New Zealand's Exclusive Economic Zone, including around the Chatham Islands and south of the Auckland Islands (Robertson et al. 2003; Thompson and Sagar 2008; ACAP 2011; Thompson et al. 2011; Torres et al. 2011), with chick-rearing birds utilising areas off the south-east coast of Australia and around Tasmania (Thompson and Sagar 2008; Torres et al. 2011). Juveniles and non-breeding adults range throughout the waters off southern Australia and South Africa (Robertson et al. 2003; Thompson et al. 2011; Petersen et al. 2008; Thompson and Sagar 2008; ACAP 2011). Juveniles and non-breeding adults have also been reported in the south-western Atlantic Ocean off Uruguay and northern Argentina (Jimenez et al. 2009; ACAP 2011).

Information from the closely-related shy albatross indicates that during the early incubation period the ratio of incubating to non-breeding birds is high as most non-breeders are at sea during the middle of the day (Barry Baker unpublished). This assumption was subsequently confirmed by observations at the South West Cape colony in November-December 2007 (Paul Sagar and David Thompson unpublished), and photographic evidence indicates the number of non-breeding birds was higher during January counts (see below).

The Site

The Auckland Islands (50^{o} 44'S, 166^{o} 06'E) lie 460 km south of New Zealand's South Island, and comprise the largest island group in the New Zealand sub-Antarctic. The archipelago consists of four larger islands (Auckland, Enderby, Adams and Disappointment islands), together with a set of smaller islands (Peat 2006). Within the archipelago, white-capped albatross breed mainly on Disappointment Island, located to the west of the main Auckland Island, with smaller colonies situated on the South West Cape of Auckland Island and on the southwest coast of Adams Island (Tickell 2000). Disappointment Island is 4 km long by up to 1 km wide, and is covered in *Poa* grassland and giant herbs, with scattered areas of shrubland and fellfield around the top of the island (Peat 2006). The island rises steeply from the sea to a plateau, with white-capped albatrosses breeding extensively on the slopes but avoiding the plateau. Birds breeding at the colonies on South West Cape and Adams Island also confine nesting to steep, tussock-covered slopes.

Field Work

Every year from 2006/07 (hereinafter 2006) to 2013/14 (2013) I chartered a single-engined Squirrel AS350B3 helicopter to conduct a return flight to the Auckland Islands group. The survey crew included a pilot, two photographers, a flight logistics manager and a New Zealand Department of Conservation representative. From 2006 to 2010 flights were conducted in December to coincide with the early incubation period of the breeding cycle. At this time it was anticipated that birds would have just completed egg laying (Paul Sagar unpublished), and hence most pairs that attempted to breed would still be attending active nests. For logistical reasons the flights for the 2011-2013 counts were undertaken in mid-January. This timing was not ideal with respect to the breeding cycle of white-capped albatross, as although hatching would not have commenced, some nests could be expected to have failed and those breeding pairs may have abandoned their breeding sites. The dates of my visits to the Auckland Islands were 16 December 2006, 13 December 2007, 14 December 2008, 3 December 2009 and 15 December 2010, 11 January 2012, 14 January 2013 and 20 January 2014.

For all flights I selected a weather window that predicted clear flying conditions with minimal low-level cloud. In all years I was able to obtain clear photographs of all colonies. Photography was timed to occur between 1100 to 1600 NZDT.

For all photography I conducted two circuits of each colony to capture the images used to count the breeding birds on the island. The survey photographs of Disappointment Island were taken at an altitude of about 400 metres. The two photographers were positioned on the port side of the aircraft to permit them to take photographs of each colony simultaneously. All photographs were taken through the open port side of the aircraft using Nikon D300 or D800 digital cameras and an image-stabilised Nikkor 70—200 mm F2.8 zoom lens, or an image-stabilised 300 mm F2.8 telephoto lens. Shutter speeds were set at 1/1000 s or faster to minimise camera shake, and every effort made to ensure that the photographs were taken perpendicular to the land surface. I took two series of photographs: 'survey photos' and 'close-up' photos. Survey photos comprised a complete series of small, overlapping images that covered the entire area of the island where albatrosses were nesting. These images were taken systematically to enable easy production of photo-montages of nesting colonies, and were generally taken with the zoom lens set at a focal length of 70 mm. If greater magnification was used, the focal length of the zoom lens was adjusted only between passes, not within each pass sequence over the colony. Close-up photos were taken with maximum photo-extension (200 mm or 300mm) to assist in determining the proportion of empty nests and non-breeding birds in the colonies.

The two photographers took approximately 3 000 digital photographs each during every survey flight. All photographs of the colony were saved as fine JPG format files, and then backed-up and stored in separate locations.

Counting protocol

I used protocols developed previously for aerial censuses of Chilean albatross colonies (Arata et al. 2003; Robertson et al. 2007) and subsequently refined in my surveys of the Auckland Islands and other sites (Baker et al. 2014). Briefly, 30 photographic montages of Disappointment Island, eight of South West Cape and one of Adams Island were constructed from overlapping photographs using the image editing software package ADOBE PHOTOSHOP (http://www.adobe.com/). The boundaries of the photographic montages for each year were generally consistent across years although slight differences between years were inevitable due to different photographic angles. Photomontages were made only of the slope habitats of Disappointment Island, South West Cape and Adams Island because white-capped albatrosses were only observed to nest on slopes. Counts of all white-capped albatrosses on each montage were then made by magnifying the image to view birds and using the paintbrush tool in PHOTOSHOP to mark each bird with a coloured circle as they were counted. To assist with counting I used MOUSECOUNT software (http://www.kittyfeet.com/mousecount.htm) and a hand-held click counter. Once all birds had been counted on a photo-montage, the file was saved to provide an archival record of the count. Each single bird attending a nest was assumed to represent an annual breeding pair (but note process for adjusting raw count data, described below). While most birds were alone at nest sites, I also counted instances when two birds were sitting close together (i.e. inside the pecking distance that defines the minimum distance between nests) and assumed both to be members of a nesting pair. In this situation, both birds were counted, and the number of pairs recorded. The number of pairs was subsequently deducted from the total number of birds to derive a raw count of annual breeding pairs.

From the photomontages it was not possible to reliably assess if a bird was a breeding bird or a loafer. For each year at least 15 close-up photos were examined to assess the proportion of empty nests and non-breeding birds in the colonies. All birds were closely examined to assess their breeding status and categorised as being either ‘on nest’, ‘not on nest’, or ‘not sure’. All visible nests were recorded as being either unoccupied (empty) or occupied. A bird standing on an empty nest was classified as being a loafer, and the nest classified as being ‘empty’. All birds recorded as being ‘on nest’ were assumed to be breeding birds, and all other birds assumed to be loafers. These data were used to calculate the proportion of loafing birds present in the colony, and the raw count for each year subsequently adjusted down to derive an estimate of annual breeding pairs for each colony. Close-up photos were not available for 2006 and the raw counts were adjusted by the mean proportion of loafers observed for all December counts (mean 0.01; 2007-2010).

Counts of photo montages in all years except 2006 were undertaken by one observer only. For 2006 I undertook multiple counts of photomontages from the December census to estimate observer variability associated with miscounting and misidentifying white spots on the ground as birds. These count data were statistically modelled by Poisson regression, a form of a Generalised Linear Model (McCullagh and Nelder 1989), with observer and area as fixed effects. Regression diagnostics and model checking indicated good model fit and there was no evidence of any significant difference ($p = 0.5$) between observers and hence no evidence of an observer bias. I have no reason to believe that data collected subsequently should have different distributional properties to my 2006 data, and so I assume the current data are also compatible with a Poisson model. Thus I present raw counts only and assume the standard deviation is estimated as the square root of the count, a property of the Poisson model. All population size estimates are presented ± 95% confidence intervals, calculated as the mean plus or minus two times the standard error.

Ground counts

There are several likely sources of bias and identifiable components of variability in using aerial survey techniques, some of which can be addressed with ground truthing, and some of which cannot (see detailed discussion below). I undertook ground counts to estimate the proportion of birds in a colony that were sitting on empty nests, and the proportion of loafers present in the colony, so that correction factors could be developed to improve the estimate of annual breeding pairs derived from aerial photography. Ground counts were undertaken within a week of the 2007 and 2008 aerial counts.

At Disappointment Island counts of occupied nests were undertaken in 2008 by two observers to determine the proportion of birds sitting on nests without an egg. All occupied nests encountered 1 m either side of a randomly placed transect were inspected and the presence of eggs recorded. These counts were undertaken on 9 December 2008 between 1200 and 1230 NZDT.

At South West Cape, Auckland Island, counts were conducted in 2007 and 2008 by three observers who independently recorded the number of birds sitting or standing on nests, the number of pairs (partners accompanying an incubating bird), and the number of non-breeding birds present in four well defined areas of the colony. Counts were made every hour between 1030 and 1630 NZDT.

Trend analysis

To assess population trend in total counts I used an appropriate Generalised Linear Model (Nelder and McCullough 1989) where the response was specified as an over dispersed Poisson distribution and the link was logarithmic. To allow for possible non-linear trend effects I used regression splines with a single knot at 2010.

Trend analyses were also run using software program TRIM (TRends and Indices for Monitoring Data; Pannekoek and van Strien 1996). TRIM is a freeware program, developed by Statistics Netherlands and is the standard tool used by the Agreement for the Conservation of Albatrosses and Petrels (ACAP) to analyse trends.

I used the linear trend model with stepwise selection of change points (missing values removed) with serial correlation and over-dispersion taken into account. Following Delord et al. (2008), I analysed overall population trend by making a log-linear regression model with Poisson error terms. Because I was interested in identifying the changes in population trends across years, I started the analysis with a model with change points at each time-point, and used the stepwise selection procedure to identify change points with significant changes in slope based on Wald tests with a significance-level threshold value of 0.01 (Pannekoek and van Strien 1996). I took into account over-dispersion and serial correlation since they can have important effects on standard errors, although they have usually only a small effect on the estimates of parameters (Pannekoek and van Strien 1996). No covariate was used. The annual population rate of change was calculated using the relationship:

$$r = \ln \lambda = \ln Nt{+}\ 1/\ Nt$$

where Nt and Nt+1 are the number of pairs breeding in year t and t + 1 respectively (taken to be the number of breeding birds counted in year t and t + 1) and λ the population growth rate (Caughley 1980). It was assumed that all the nesting birds were detected. Nt+1, Nt and λ were given by TRIM.

TRIM classifies trends by converting the multiplicative overall slope estimate in TRIM into one of the six categories shown below. The category depends on the overall slope as well as its 95% confidence interval.

Strong increase - increase significantly more than 5% per year (5% would mean a doubling in abundance within 15 years). Criterion: lower limit of confidence interval > 1.05.

Moderate increase - significant increase, but not significantly more than 5% per year. Criterion: 1.00 < lower limit of confidence interval < 1.05.

Stable - no significant increase or decline, and it is certain that trends are less than 5% per year. Criterion: confidence interval encloses 1.00 but lower limit > 0.95 and upper limit < 1.05.

Uncertain - no significant increase or decline, but not certain if trends are less than 5% per year. Criterion: confidence interval encloses 1.00 but lower limit < 0.95 or upper limit > 1.05.

Moderate decline - significant decline, but not significantly more than 5% per year. Criterion: 0.95 < upper limit of confidence interval < 1.00.

Steep decline - decline significantly more than 5% per year (5% would mean a halving in abundance within 15 years). Criterion: upper limit of confidence interval < 0.95.

Results

Aerial and ground counts

Count data over eight years show strong inter-annual fluctuations (Table 1), a characteristic observed for many other seabird species (e.g. Congdon et al. 2007). By far the majority of the Auckland Islands population was breeding on Disappointment Island, where raw counts ranged from 70 569 pairs in 2009 to 110 649 pairs in 2006 over the 8 years of the study. At South West Cape, numbers ranged from 4 161 to 6 548, and at Adams Island from 79 to 215 (Table 1).

Analysis of close-up counts indicated that there were few non-breeding birds in the colony when counts were carried out in 2006-2010 (December, early incubation), but in 2011-2013 (January, late incubation period) more non-breeders were present. Across four years of close-up counts from 2007-2010, 3 939 of the 3 993 visible birds (99%) were sitting on nests (Table 2). Close-up counts from 2011-2013 indicated that 2 963 of the 3 456 visible birds (86%) were sitting on nests (Table 2). Thus the proportion of non-breeding birds during the last three years (January, late incubation) ranged from 7-22%. These differences were taken into account when assessing population trends.

Also apparent in the close-up photographs were large numbers of empty nests (Table 2). For the seven years 2007 to 2013 I counted a total of 2 777 empty nest pedestals and 6 902 occupied nests (29% empty) in randomly selected close-ups photographs.

Ground counts of nests inspected on Disappointment Island on 9 December 2008 (early incubation) showed that 447 occupied nests (93.5%) contained eggs and 31 (6.5%) were empty. At South West Cape ground counts in 2007 and 2008 (December, early incubation) confirmed the impression provided by the close-up photos that few non-breeding birds are generally present in the colony during December at the time of day that the aerial photography was undertaken. From 84 observations, ≤ 2% of birds present were non-breeders on 86% of observations, and ≤ 5% on 97% of the total observations. The maximum number of non-breeders present at any one time was 10%, recorded at 1630 NZDT, and later in the day than when aerial counts were undertaken.

Estimated annual counts for all three breeding sites in the Auckland Islands were adjusted to account for the presence of loafing birds, giving adjusted estimates of annual breeding pairs for each year from 2006 to 2013 (Table 1). The subsequent mean number of annual breeding pairs in the Auckland Islands during this period was 90 141, with annual estimates ranging a 73 838 to 116 025.

Trend analysis

Trend analysis for all sites combined using regression splines showed no clear evidence for monotonic decline over the 8 years of the study. Given this I do not have sufficient evidence to reject the null hypothesis of no trend in the total count of annual breeding pairs (Figure 1).

Table 1. Raw counts with 95% confidence intervals of white-capped albatrosses in the Auckland Islands in December 2006-2010 and January 2012-2014, and estimated annual breeding pairs following adjustment to account for the presence of non-breeding birds.

Island		**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**
<u>Raw counts</u>									
Adams		**-**	**79**	**131**	**132**	**117**	**178**	**215**	**184**
	LCI		61	108	109	95	151	186	157
	UCI		97	154	155	139	205	244	211
Disappointment		**110 649**	**86 080**	**91 694**	**70 569**	**72 635**	**93 752**	**111 312**	**89 552**
	LCI	109 984	85 493	91 088	70 038	72 096	93 140	110 645	88 953
	UCI	111 314	86 667	92 300	71 100	73 174	94 364	111 979	90 151
SW Cape, Auckland		**6 548**	**4 786**	**5 264**	**4 161**	**4 370**	**5 846**	**6 571**	**5 542**
	LCI	6 386	4 648	5 119	4 032	4 238	5 693	6 409	5 393
	UCI	6 710	4 924	5 409	4 290	4 502	5 999	6 733	5 691
Total Auckland Islands		**117 197**	**90 945**	**97 089**	**74 862**	**77 122**	**99 776**	**118 098**	**95 278**
	LCI	116 512	90 342	96 466	74 315	76 567	99 144	117 411	94 661
	UCI	117 882	91 548	97 712	75 409	77 677	100 408	118 785	95 895
<u>Adjusted counts</u>									
Proportion non breeding birds		0.01	0.01	0.01	0.01	0.01	0.07	0.13	0.22
Annual pairs		116 025	90 036	96 118	73 838	76 119	92 692	102 273	74 031

Table 2. **Summary of counts of randomly selected close-up photographs taken each year at Disappointment Island in December 2007-2010 and January 2012-2014.**

	Month	On Nest	Not sure	Not on nest	Total Birds - breeding status known	Proportion of non-breeding birds	Empty nests	Total nests
2007	**December**	805	21	4	809	1%	326	1 131
2008	**December**	1 590	20	29	1 619	2%	438	2 028
2009	**December**	937	23	13	950	1%	633	1 570
2010	**December**	607	16	8	615	1%	343	950
2011	**January**	1 007	31	77	1 084	7%	291	1 298
2012	**January**	1 096	63	169	1 265	13%	n/a	663
2013	**January**	860	24	247	1 107	22%	504	1 364
Totals		**6 469**	**166**	**494**	**7 449**	**7%**	**2 777**	**9 679**

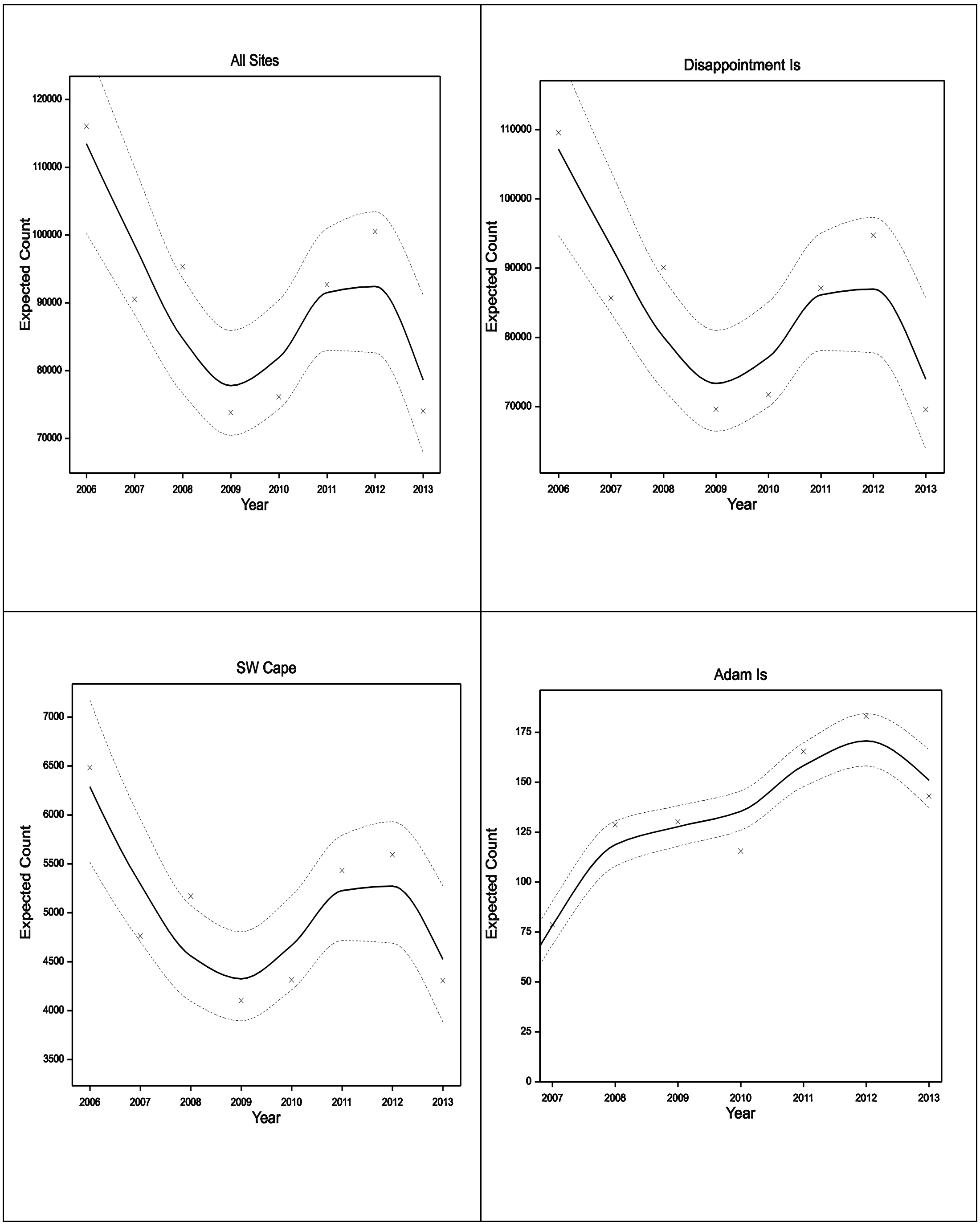

Figure 2 Data points (total counts as adjusted for the presence of non-breeding birds), regression trend line with associated 85% confidence intervals for annual breeding pairs of white-capped albatross at three sites in the Auckland Islands. Non-overlap of the 85% CI between any two points infers significance at P=0.05. Note that scale differs on the Y axis.

Using TRIM for all sites combined and analysing eight years of adjusted count data (2006 to 2013 breeding seasons), the stepwise procedure stepwise procedure for selection of change points indicated significant change points in all years ($p < 0.01$ for Wald tests). The population size estimates computed from the model indicate an average growth rate of -3.16% per year ($\lambda = 0.9684 \pm 0.001$; assessed by TRIM as moderate decline.

Discussion

Annual counts

My study provides the first complete estimate of the number of annual breeding pairs and population trend of white-capped albatrosses at the Auckland Islands. When the raw counts are adjusted for the presence of loafing birds, the estimated number of annual breeding pairs over the last eight years has ranged from a high of 116 025 annual breeding pairs in 2006 to a low of 73 838 in 2009, with mean estimated number of annual breeding pairs of 90 141. These numbers exceed the early published estimates of Gales (1998) and Taylor (2000) (70 000 and 70-80 000 pairs, respectively).

Two aspects of the counts were notable. Firstly, the count patterns at both Disappointment Island and South West Cape were very similar in every year (Figure 1). This would not be necessarily expected, and provides evidence that both populations are impacted equally by the same ecological and environmental influences. Under resource constraints, annual monitoring of the smaller South West Cape colony could serve as a useful proxy for the population as a whole. Secondly, the number of loafers in the colonies increased greatly as incubation progresses, as has been observed for other albatross species (Tickell 2000). Evidence from the close-up photographs across eight years indicates that the number of loafing white-capped albatrosses at Disappointment Island was very low early in the incubation period (<2% for all December counts), but higher later in the breeding season (7%, 15% and 22.3% for January counts in 2011, 2012 and 2013, respectively). This has implications for the timing of future counts, as it is desirable that this parameter is measured each year, particularly when counting is undertaken later in the breeding season. While acknowledging that helicopter availability in this remote locality is always likely to have some influence on the timing of future counts, it is recommended that future counts are timed for mid-December, if logistically feasible.

Trend analysis

Population size estimates computed from the TRIM model indicate an average growth rate of - 3.16% per year ($\lambda = 0.9684 \pm 0.001$); assessed by TRIM as moderate decline. I note, however, that a simple linear trend analysis, as performed by TRIM is not well suited to a data set with high inter-annual variability. Trend analysis using regression splines is more appropriate to such data sets, and showed no evidence for monotonic decline over the eight years of the study, therefore providing insufficient evidence to reject the null hypothesis of no trend in the total population. Thus the current population trend is uncertain and more data is necessary before the population status of the white-capped albatross can be determined.

Sources of error in photographic census, and the value of ground-truthing

As for any wildlife survey method, aerial photography must contend with sources of sampling error. The observed strong inter-annual fluctuations in the count observed in this study would encompass counting error, the presence of loafing birds during counts, environmental stochasticity and other unknown variables that are not easily quantified.

Ground-truthing has been used in other photographic censuses of albatross colonies to estimate the bias associated with birds loafing in colonies, birds sitting on nests without an egg, and to identify areas where nests may be obscured from the air by topographical features (Robertson et al. 2007). The information gained from ground surveys has then be used to estimate the total number of breeding pairs from the total number of birds counted.

I identified the following likely sources of bias and identifiable components of variability in using aerial survey techniques, some of which can be addressed with ground truthing, and some of which cannot. These include:

(1) The total number of active nests will be overestimated due to the presence of loafing birds and birds sitting on nests without eggs. For black-browed albatross colonies in Chile, Robertson et al. (2007) estimated that nearly 12% of birds attending a colony fell into one of these two categories. Simultaneous ground-truthing revealed that 5% of the birds photographed were loafing in the colony and a further 7% were sitting on empty nests. The size of these errors would differ depending on the time of day and stage of breeding that surveys were conducted.

I chose to use the evidence on the proportion of loafers present in colonies, derived from close-up photographs and ground counts to deflate the raw counts to estimate the number of annual breeding pairs. However, it could be equally valid to inflate counts to some degree using this data, as a loafer may be a failed breeding bird, particularly so when birds are sitting on nests without eggs. As such, my estimates of annual breeding pairs should be considered to be conservative. I also recognise that ground-truthing data assessing the proportion of birds sitting on empty nests will not reliably provide a correction factor relevant to determining annual breeding pairs, as a bird sitting on an empty nest may have laid and subsequently lost its egg, may be yet to lay, or simply be a non-breeding loafer.

(2) Differences between observer counts will generate variability in the count, as will misidentification of birds in mixed species colonies. Fortunately, my analyses suggest that the error associated with my counts was no larger than the intrinsic error expected in count data, and there were no other species nesting amongst the white-capped albatross colonies.

(3) Poor stitching of the photographs will generate variability in counts. Omission or double-counting of albatrosses near stitch lines due to parallax has been considered a problem in other studies (Robertson et al. 2007). For the counts at all breeding sites in the Auckland Islands the nature of the terrain was such that I am confident that on most stitch lines no such errors occurred. On most images the ridge lines were easily defined and I am confident that birds were not missed or double counted. Where it was difficult to draw these lines any error would not have exceeded two hundred birds in total across all stitched images in any year.

(4) Ground-truthing may permit identification of 'detection error' in areas where nests may be obscured from the air by topographical features such as jumbled rock substrate, but this is unlikely to have been a problem for the Auckland Island sites. Note however, that in some cases where site topography is uneven, it is possible to miss small colonies in ground counts that may be readily observed from the air (Robertson et al.2007; G. Robertson unpublished).

While ground-truthing may improve the accuracy of population estimates derived from aerial surveys, it needs to be recognised that the timing of aerial and on-ground counts needs to synchronous if meaningful correction factors are to be developed. In any albatross colony, nests fail after laying as eggs are broken or become buried in the mud-nest pedestals. In the closely related shy albatross, some birds may continue to attend nests for some time after eggs are lost or broken. However, as the time-lag between an aerial and on-ground count increases, the relativity between estimates derived from both counts is likely to decrease. Access to many sub Antarctic islands is often difficult for both logistic and financial reasons, and the uncertainty associated with access may provide a valid reason to rely on aerial counts for estimating population size at sites where it is feasible to do so. As advocated by Robertson et al. (2007) and used by Arata et al. (2003) and in this study, the use of high resolution digital photographs and subsequent magnification on a monitor to enhance the images of individual birds, can provide improved information on posture and behaviour that may enable breeding birds and loafers to be separated. Elimination of ground-truthing has further benefits in reducing disturbance at nesting colonies.

Despite the strong inter-annual fluctuations, the data are useful for tracking change in the white-capped albatross population since they have been collected at roughly the same time of the breeding cycle (incubation), allowing inferences about long-term trends to be made. This information should provide a statistical basis for making decisions about management of these populations.

I am confident that the observed differences in counts are real and not an artefact of technique, although the timing of the counts over the last three years (2011-2013) of the study differed by one month from all previous counts. Any bias that exists around the counts due to the presence of loafers should be consistent across all years when counts are undertaken at the same time of year, as the close-up data indicates. Adjusted counts based on these data deal effectively with bias if counts are undertaken at different times in the breeding season.

In all other aspects, the methods employed and the personnel used for the photography, construction of photo montages and counting were essentially identical for all years. It is also clear from an analysis of the close-up photographs taken in all but the first year of the study that there were a number of visibly unoccupied nest pedestals across the two larger colonies. Such a high proportion (0.29) of empty to occupied nests is usually not apparent in colonies of the medium to small albatrosses until later in the breeding season. Also apparent is an increase in the number of non-breeding birds present in the colony in counts taken later in the breeding season (January), which was evident from the 'close-up' photos and on-ground observations (David Thompson unpublished).

There are two possible explanations for the different totals estimated between years. White-capped albatrosses are now considered to be biennial breeders, as recent research has indicated (Paul Sagar and David Thompson unpublished). As such, I would expect to see larger inter-annual fluctuations in counts than that typically observed with annual breeding species where populations are less variable between years (Tickell 2000). Breeding may have commenced earlier in some years, placing my counts at a time after significant early nest failure may have occurred, although this is unlikely given the synchronous nature of breeding in albatrosses (Tickell 2000). Certainly, it needs to be remembered that counts over the last three years (20112013) have been made a month later than in the first five years of counts. While I have taken into account the presence of more non-breeding birds in the colony in the last 3 years, I would also expect numbers later in the season to be lower than those recorded at the end of egg laying (December) as some pairs would have failed and ceased attending the colony.

It is also possible that the differences between years represent normal inter-annual variation in breeding, with variable resource availability in later years causing many birds to skip breeding in those years. A further possibility — that we are observing a population decline — seems unlikely with the evidence that trend analysis showed no evidence for monotonic decline over the eight years of the study.

An increasing number of studies in recent years (reviewed in Phillips et al. 2016) have focused on potential impacts on seabirds of climatic variation, including annual sea surface temperature (SST) and marine productivity, and global cycles (El Niño Southern Oscillation, North Atlantic Oscillation). Warmer conditions (higher SST, especially at foraging grounds) usually have negative effects on demographic parameters, especially breeding success, although the relationships can be non-linear. In contrast, black-browed albatrosses from Kerguelen benefited from increased SST, with evidence for contrasting responses to conditions in breeding vs. non-breeding areas (Phillips et al. 2016). Although juvenile survival can be reduced under warmer conditions, there is little evidence for a comparable effect on adult survival in albatrosses and petrels. Modelling suggests that responses to future climatic change will be species-specific, with few impacts predicted for northern species but steep declines for species in the Southern Ocean as a consequence of increased SST and decreased sea ice extent (Phillips et al. 2016). There have also been shifts in distribution and breeding phenology of seabirds in response to climate change (Peron et al. 2010, Weimerskirch et al. 2012; Phillips et al. 2016).

Conservation implications

The remoteness of breeding sites and difficulty of access has previously constrained development of a comprehensive estimate for size of the breeding population of white-capped albatross (Croxall and Gales 1998, Taylor 2000). While attempts have been made at times over the last 20 years to conduct counts at Disappointment Island and South West Cape, where the bulk of the global population breeds, details of these have never been published and it is difficult to assess the methodology used, the time of year counts were made, the completeness of the counts, and any population trend beyond the data I have collected.

With only the reputedly small colony on Bollons Island (Gales 1998; Tennyson et al. 1998; Robertson 1975) not counted in this study, my estimates represent the first reliable population estimate for this species. These estimates indicate that global population is currently c.90 000 annual breeding pairs, which is much larger than previously thought. This may be the result of sustained population growth since the 1970s, or simply reflect inaccuracy of the earlier counts in a population that is stable.

In the global review of fisheries-related mortality of shy and white-capped albatrosses described in Chapter 1 I estimated that 8 000 white-capped albatrosses were killed each year as a result of interactions with commercial fisheries in the Southern Ocean. This level of mortality highlights the need to continue to acquire accurate population estimates and trends for white-capped albatross populations to assess the impact of fisheries operations on this species. Although annual counts over the study period indicate the population may be stable, population trend is still uncertain, and ongoing population monitoring is recommended to clarify the population’s status and determine if current levels of fishing mortality are sustainable.

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Chapter 3

Population viability analysis of shy (*Thalassarche cauta*) and white-capped (*T. steadi*) albatross.

Abstract

Shy and white-capped albatrosses, *Thalassarche cauta* and *T. steadi* respectively, are closely-related and phenotypically similar species that are known to be impacted by fisheries-related mortality across a large area spanning Australian, New Zealand and South African waters. I conducted a Population Viability Analysis for both these species to understand the key demographic factors driving population trends and facilitate the evaluation of current levels of fisheries-related mortality. The models developed indicated that the estimated bycatch from global fisheries is unsustainable for both species. However, as the observed population trend for both these species over the last 10-20 years has not shown the rate of decline predicted by modelling, it is likely that the bycatch estimates for both species have been over-estimated. The Potential Biological Removal level calculated for white-capped albatross and used in current risk prioritisation showed negative population growth, a consequence of being based on a recovery factor F_R of 1. Application of a PBR with $F_R = 0.1$ or $F_R = 0.2$, as recommended in the literature for threatened species, would be appropriate for both species, and led to positive population growth when applied to my base model for both species. While the estimated levels of bycatch in this study were largely found to be unsustainable, assessing and minimizing bycatch should be a priority, irrespective of population-level impacts. Application of PBRs to manage bycatch can be effective in maintaining populations at prescribed levels, but focus also needs to be maintained on the development and implementation of measures to mitigate the bycatch of albatrosses and other seabirds. Finding solutions to bycatch problems requires a mix of legislative and political measures, combined with sound science to define problems and develop technological answers.

Introduction

Seabirds are caught incidentally in many commercial fisheries using various gear types throughout the world, and concerns have arisen about the clear link between incidental catch and observed population declines for many seabird species (Weimerskirch and Jouventin, 1987; Croxall et al., 1990; de la Mare and Kerry, 1994; Prince et al., 1994; Weimerskirch et al., 1997; Gales, 1998; Weimerskirch and Jouventin, 1998; Tuck et al., 2001; Nel et al., 2002; Terauds et al. 2006). Global longline, trawl, gillnet, and purse-seine fisheries have been identified as fisheries of potential concern (Gales 1998; Baker and Wise 2005; Baker et al. 2007; Anderson et al. 2011; Zydelis et al. 2013).

Management of fisheries in a contemporary context requires that managers not only focus on fishing impacts of target species, but also take into account impacts on non-target species. The Code of Conduct for Responsible Fisheries (FAO, 1995) promotes, among other things, the minimization of catch of non-target species, the protection of endangered species, and the assessment of the impacts of environmental factors on target stocks and species belonging to the same ecosystem. The Code also encourages the adoption of a precautionary approach, taking into account uncertainties relating to the size and productivity of target and non-target stocks, levels and distribution of fishing mortality and the impact of fishing activities on non-target and associated or dependent species. Under the Code, an International Plan of Action (IPOA) relating to the incidental catch of seabirds in longline fisheries (IPOA-Seabirds) (FAO, 1998) has been developed, and subsequently expanded to include other fishing gears (FAO 2009).

The implementation of the IPOA is voluntary but is widely accepted as good practice (Lack 2007). The IPOA-Seabirds stipulates that countries with fisheries, or a fleet that fishes elsewhere, should carry out an assessment of these fisheries to determine if a seabird bycatch problem exists and, if so, to determine its extent and nature. If a problem is identified, countries should adopt a National Plan of Action - Seabirds (hereafter NPOA-Seabirds). While national seabird assessments can be useful and effective for species and populations with restricted distributions, they are insufficient for determining appropriate management actions for taxa with distributions that extend beyond national boundaries. What is required for such migratory species are global assessments of bycatch and the potential impact on population growth (Lewison et al. 2005; Baker et al. 2007).

Shy and white-capped albatrosses, *Thalassarche cauta* and *T. steadi* respectively, are closely-related and phenotypically similar species (Abbott and Double, 2003 a, b; Double et al., 2003). Shy albatrosses breed in Australia on three islands around Tasmania, whereas white-capped albatrosses breed mainly on islands in the Auckland Islands group in New Zealand’s subantarctic. Their global population sizes are estimated at 15 000 and 90 000 annual breeding pairs, respectively (Alderman et al. 2011; Chapter 2). Both species are known to be impacted by fisheries-related mortality across a wide spatial scale, including in Australian, New Zealand and South African waters (Ryan et al. 2002; Abbott et al 2006; Baker et al. 2007; Richard and Abraham 2014).

While there has been no population viability analysis of shy albatross populations, differing opinions exist about estimated levels of bycatch, as well as the capacity of the white-capped albatross population to sustain the impact of fisheries bycatch. In Chapter 1 (Baker et al. 2007) I conducted a global assessment of the effect of fisheries on both shy and white-capped albatrosses (collectively ‘shy-type albatrosses’) based on fisheries observer data, and estimated that a total of 8 500 ‘shy type albatrosses’ may be killed annually, although the reliability of this estimate was low. While comparatively few shy albatrosses were killed each year, I considered that the estimated loss of over 8 000 white-capped albatrosses may be unsustainable. Subsequently, Francis (2012) estimated that the global bycatch of white-capped albatrosses, including cryptic mortality[2], exceeded 17 000 birds per year, which he believed could present a risk to population viability. Francis (2012) also estimated adult survival (based on mark-recapture data) to be 0.96 (0.91-1.00, 95% C.I.) which suggests that adults may not be overly impacted by fisheries bycatch. Richard and Abraham (2014), in an assessment of the risk of commercial fisheries to New Zealand seabirds, used a modified Potential Biological Removal (PBR) method (Wade 1998), to calculate a mean annual PBR value of 4 040 (2 590-6 340, 95% C.I.). The PBR index (Wade 1998) was originally developed under the United States Marine Mammal Protection Act to assess the maximum level of human-induced mortality that a marine mammal population can incur, while being able to stay above half its carrying capacity in the long term. This concept has subsequently been extended to seabirds (Dillingham and Fletcher 2008) and other organisms.

Richard and Abraham (2014),also calculated the total annual potential fatalities (APF) in all New Zealand trawl, longline, and gillnet fisheries to be 4 410 (2 800-6 540), suggesting that mortality levels in New Zealand alone could be placing the species at risk of declining.

The uncertainties surrounding the potential levels of bycatch for both species, together with the potential effects of spatio-temporal changes in fishing effort, provide strong incentives for further assessment of fisheries impact on the viability of both shy and white-capped albatross populations. The Population Viability

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[2] *Defined as 'birds that are fatally injured in an encounter with fishing effort but are not counted by observers because they are not recovered onboard the fishing vessel'*

Analysis conducted here provides important insights in understanding the key demographic factors driving population trends and enables the realistic evaluation of current levels of fisheries-related mortality.

Methods

Breeding sites and biology

Shy albatrosses are endemic to Australia, breeding only on three islands around Tasmania: Albatross Island (5 200 pairs); the Mewstone (9 500 pairs) and Pedra Branca (170 pairs) (Alderman et al. 2011). The total population is estimated to be 55 000 – 60 000 individuals (Alderman et al 2011). The species has been well studied and a long-term demographic study, in place since 1980 on Albatross Island, has ensured that vital rates are well estimated (Alderman et al. 2011). Breeding is annual; females lay a single egg in September each year, with chicks hatching in December and fledging in April (Gales 1993; Hedd and Gales 2005). After departing their colonies, young birds spend at least two years at sea before returning to their natal colony. Shy albatrosses have been recorded breeding at five years of age, but the majority of individuals commence breeding at eight or nine years old (Hamilton 2003; Alderman et al. 2011). Foraging range is comparatively restricted with adults remaining near their breeding colonies all year (Hedd et al. 2001; Hedd and Gales 2005; Alderman et al. 2011). Juveniles and immatures also tend to remain in southern Australian waters, although they do demonstrate a more extensive distribution, with some individuals from the Mewstone colony traversing the Indian Ocean and foraging in South African waters (Abbott et al. 2006; Alderman et al. 2010; Alderman et al. 2011).

White-capped albatrosses are endemic to New Zealand, breeding mainly on the Auckland Island group at three colonies (Disappointment Island - 85 000 pairs; Adams Island - 35 pairs; and main Auckland Island - 5 000 pairs) (Chapter 2), and Bollons Island (50–100 pairs) in the Antipodes Island group (Taylor 2000). The population is estimated to comprise about 425 000 (317 000–600 000, 95%CI) individuals (Richard and Abraham 2014). The breeding frequency and season for this species are poorly known, with most information gained from a five-year study of c.70 nesting pairs at the South West Cape, Auckland Island colony (Thompson et al. 2011). Egg-laying commences in mid-November, with hatching underway by mid-January, extending into early February, and chicks fledging in June (Thompson et al. 2011). White-capped albatross appear to have a breeding strategy intermediate between annual and biennial (Thompson et al. 2011). Francis (2012) reported the probability that a bird that bred in one year would also breed in the next year to be 0.63, whereas the probability that a bird that didn't breed in one year but which would breed in the next year was 0.78.

White-capped albatrosses forage extensively across the Tasman Sea, around south-eastern Australia when breeding. At other times the majority of birds remained in Australasia year-round (Thompson et al. 2011). Geolocation data from 25 birds showed that 24% of birds remained close to New Zealand and the eastern Tasman Sea year-round, 40% moved as far as Tasmania and south-eastern Australia, 20% moved westwards to southern and south-western Australia, while the remaining 16% of birds migrated to the south and south-western coasts of South African and Namibian waters (Thompson et al. 2011).

Modelling

Population models were developed based on the most robust population dynamics data for white-capped and shy albatrosses. I used software program VORTEX (Version 10; Lacy and Pollak 2014) as it is widely used for undertaking PVAs in a range of situations (e.g. Prowse et al. 2013; Midwood et al., 2014, Hamilton and Baker 2015). This program simulates survival and reproductive events in successive years for each individual in a population by the Monte Carlo method. It is stochastic in that it imposes variations in annual survival and reproduction by random number generations according to prescribed probability distributions for reproduction and survival rates.

PVA models were developed using data from the Albatross Island study (Hamilton 2003; Alderman et al. 2011) for shy albatross, and from the limited study at South West Cape (Francis 2012; Thompson et al. 2012) for white-capped albatross. Where suitable data were not available, estimates of population parameters from research carried out on congeners of similar size were used, principally the well-studied Buller's albatross *T. bulleri* (Francis and Sagar 2012). The input demographic parameters are summarised in Table 1 with further information below.

Table 1: White-capped albatross and Shy albatross demographic parameters and incidental fisheries capture levels used in the Population Viability Assessment (PVA). Further details of how parameters were derived are in Methods.

Parameter	White-capped Base Model	Shy Base Model
Inbreeding Depression	None	None
EV concordance of Reproduction & Survival	None	None
Breeding strategy	Long-term monogamy	Long-term monogamy
Young per year	1	1
Female breeding age (years)	8** (Hamilton, 2003)	8 (Hamilton 2003; Alderman et al. 2011)
Female maximum breeding age (years)	29** (Hamilton, 2003)	29 (Hamilton, 2003)
Male breeding age (years)	8** (Hamilton, 2003)	8 (Hamilton, 2003)

Parameter	White-capped Base Model	Shy Base Model
Mean % adult females producing progeny/year (EV = environmental variation)	Estimated at 68% (EV=10) (Francis 2012), adjusted to 75% to achieve positive population growth	87% (EV=4) (Hamilton, unpublished)
Maximum life span	40 (assumed)	40 (assumed)
Sex ratio at birth (males)	50:50 (assumed)	50:50 (assumed)
Density Dependant Reproduction	No	No
% males in breeding pool	100% (assumed)	100% (assumed)
Breeding success	0.63 (0.51-0.75) (Francis 2012)	0.46 (Alderman et al. 2011)
Female & Male mortality (%) (SD)		
Age 0	74.5 (4)	77 (4)
Ages 1-7	4 (2)	4 (2)
Adults 8+	4 (2)	4 (2)

Parameter	White-capped Base Model	Shy Base Model
Initial population size	424 000 in 2014 (derived from Richard & Abraham 2014)	60 000 in 2009 (derived from Alderman et al. 2011)
Carrying capacity ****	700 000	100 000
Functional extinction at half the population size	212 000	15 500
Additional scenario options added to the Base Models:		
Harvest (Annual mortality in fisheries)	(i) 11 500 birds/year	(i) 1 950 birds/year
	(ii) 6 780 birds/year	(ii) 940 birds/year
	(iii) 8 887 birds/year (estimated PBR*** R_F = 1.0)	(iii) 1 524 birds/year (estimated PBR*** R_F = 1.0)
	(iv) 4 444 birds/year (estimated PBR*** R_F = 0.5)	(iv) 762 birds/year (estimated PBR*** R_F = 0.5)
	(iv) 1 777 birds/year (estimated PBR*** R_F = 0.2)	(iv) 305 birds/year (estimated PBR*** R_F = 0.2)

Parameter	White-capped Base Model	Shy Base Model
	(iv) 889 birds/year (estimated PBR*** R_F = 0.1))	(iv) 152 birds/year (estimated PBR*** R_F = 0.1)

*Southern Buller's albatross data used as a proxy following Richard & Abraham 2014

**Shy albatross data used as a proxy for White-capped albatross

***Potential Biological Removal level (Wade 1998); R_F = recovery factor

****Carrying capacity was purposefully set to a limit well above that which would be achieved by natural population growth to eliminate any impact on the model

Each model was run for a 30 year period with 2000 simulations, with quasi-extinction defined as a 50% reduction in population size. Quasi-extinction time, defined here as a 50% reduction in population size, was used as a reference point to describe a population's state and define critical levels of fishing on a population (Baker and Wise 2005).

The mortality estimates modelled were based on the inverse of published survival estimates (Alderman et al. 2011, Francis 2012). The estimated adult mortality of 4% for both species is a low value and indicates that any impact of fishing is likely to be minimal for this age class. Mortality estimates for juvenile and immature white-capped albatross were not available and so I used those estimated for shy albatross (Alderman et al. 2011). As these published estimates were based on band resights they intrinsically included existing levels of fisheries mortality as well as mortality from other causes. I therefore adjusted these or other vital rates to account for the inclusion of fishing-related impacts (Table 2) and ensure that base models projected population trajectories exhibiting positive population growth. For white-capped albatross this was achieved by adjusting the proportion of adult females producing progeny/year from my estimate of 68% (Francis 2012) to 75%.

Density dependence and inbreeding depression were not included in the models as there is no published evidence for this in either species. Both species were modelled as single populations, due to the limited information on movement of breeding females between colonies and the likelihood that there are few intra-specific ecological differences of consequence (Thompson et al. 2011). Although differences exist in the foraging range of shy albatross from different colonies (Alderman et al. 2011) these were not able to be easily incorporated into the models.

The impact of fisheries bycatch was modelled through the 'Harvest' option in VORTEX. Details of bycatch impacts modelled are provided in the subsection on fisheries interactions below.

Fisheries interactions

In Chapter 2 I conducted a global assessment of fisheries bycatch on both shy and white-capped albatrosses based on fisheries observer data. I estimated that a total of 8 500 'shy type albatrosses' may be killed annually, although the reliability of this estimate was low. This work was completed in 2007. As fisheries effort varies considerably over time in response to changing economic and biological factors, I updated these data using current knowledge on levels of bycatch from the data sources identified in Table 2. There are five areas where updated estimates require clarification:

1. The 2007 bycatch estimates were based on levels of observed and reported bycatch and, with a few exceptions, made no allowance for cryptic mortality. As such, they are likely to be substantial under-estimates of mortality (Francis 2012). Observer records of birds caught may underestimate the total number of fisheries-related fatalities as birds killed may not be observed (Richard and Abraham 2014). In longline fisheries birds may be killed but fall off hooks during the soak and gear retrieval (Brothers et al. 2010). In trawl fisheries birds may be killed when striking warps and third wires and not retained in the net (Watkins et al. 2008; Abraham 2010; Parker et al. 2013). To account for this when assessing fishery impacts on seabirds, Richard and Abraham (2014) developed cryptic mortality multipliers that were gear and species specific. Relevant values for white-capped albatrosses are 2.08 and 8.23 for longline and trawl gears, respectively (Richard and Abraham 2014). I have used these values to estimate levels of cryptic mortality for both shy and white-capped albatrosses in each fishery (Table 2) where updated estimates have not accounted for this. The trawl multiplier is high and its use controversial given the uncertainties associated with this parameter, so I also used a lower multiplier of 3.0 to adjust bycatch estimates, following advice from a fisheries observer with considerable experience in Falkland Island trawl fisheries (Ben Sullivan, unpublished).

2. Watkins et al. (2008) analysed seabird bycatch in the South African deep-water hake (*Merluccius* spp.) trawl fishery and estimated that 7 380 'shy type' albatrosses were being killed each year in the fishery. This estimate included an allowance for warp strikes (cryptic mortality), and exceeded my earlier estimate of 5 000 birds per year (Chapter 1). Subsequently, Maree et al. (2014) reported the mortality of albatrosses in this fishery had been reduced by > 95%, and in 2010 was estimated at 83 (95% CI 38–166) albatrosses. This followed the rigorous adoption of mitigation measures and a 50% reduction in annual fishing effort from 2004 to 2010. Following the approach adopted in Chapter 1, I have assumed that 29 (35%) of these birds will have been 'shy-type' albatrosses, and that most (95%) were white-capped albatross (Table 2).

3. BirdLife (2013a, 2013b) have recently estimated bycatch in both longline and trawl fisheries in Namibia. Although bycatch of seabirds is very high in both fisheries, 'shy type' albatrosses do not appear to be impacted. Accordingly, these fisheries were not considered relevant for inclusion in models of 'shy type' albatrosses described here.

4. For the multi-sectored Australian Southern and Eastern Scalefish and Shark Fishery (SESSF), in Chapter 1 I considered that only the Scalefish Hook Sector was likely to impact 'shy-type' albatrosses. Phillips et al. (2010) analysed observer data from Commonwealth Trawl fisheries operating in southern Australia and the Great Australian Bight. They estimated 861 'shy-type' albatrosses were killed in Australian Commonwealth trawl fisheries in 2006. This estimate was an extrapolation based only on net captures or birds retained on splices, and would therefore likely underestimate the total number of birds killed (Alderman et al. 2011). Fishing effort in the fishery has declined by 30% since 2006 (Penney et al. 2013) and I therefore estimate annual bycatch of 'shy type' albatrosses to be 603 birds. Again, following Abbott et al. (2006), the ratio of shy to white-capped albatrosses was assessed as being 0.35/0.65 (Table 2).

5. 'Shy type' albatrosses have been recorded in both the south Atlantic and eastern Pacific (Tickell 2000; Phalan et al. 2004), but sightings are uncommon and these areas are not considered to be part of their usual distribution (Tickell 2000; Robertson et al. 2003b). At the time of writing Chapter 1 (2007), 'shy-type' albatrosses had not been reported as bycatch amongst the birds caught in South American fisheries. Since then, Jimenez et al. (2015) reported that 29 birds had been caught incidentally by pelagic longliners fishing in Uruguayan waters between 2008 and 2011. Genetic analysis showed that 28 of these birds were white-capped albatross, with the remaining bird potentially being a shy albatross, although the genetic test was not definitive (Jimenez et al. 2015). The number of white-capped albatross caught annually in the Uruguayan longline fishery is small (Jimenez et al 2012, 2015) and conservatively estimated here at 10 birds, all assessed as white-capped albatross.

The ages of bycatch birds would appear to vary strongly by area and fishery in some cases, while remaining uncertain for other fisheries. Ryan et al. (2002) and Abbott et al. (2006) reported that approximately 44% of shy-type albatrosses caught in South Africa waters were adults. However, in the South African pelagic longline fishery only 11% of the bycatch was adults (Petersen et al. 2009). Of the 32 white-capped albatross caught off Tasmania 12 (37%) were adults (Abbott et al. 2006). Francis (2012) reported that most (92%) of the 1 135 white-capped albatross killed in New Zealand fisheries between 1996/97 to 2008/09 were adults). In Uruguay most birds seen are immatures (Jimenez et al. 2009). The ages of birds caught in the Asian Distant Water Longline Fishery have not been reported. Because of the uncertainty in the age classes of both species impacted by all fisheries, I apportioned bycatch birds equally across all nine age classes (Ages 0 to 8) and both sexes when developing modelling scenarios to assess fisheries impact.

The use of the Potential Biological Removal index was developed by Wade (1998) to assess the maximum level of human-induced mortality that a population can incur while being capable of staying above half its carrying capacity in the long term

(Richard and Abraham 2014). The purpose of the PBR is to set limits of mortality for populations with levels of human-caused mortality that may be too high, so that populations can be managed sustainably. A key factor of PBR estimates is the recovery factor R_F which ensures the recovery of populations to optimum population levels. Dillingham and Fletcher (2008) suggested that it may be reasonable to set a recovery factor $R_F = 0.5$ for 'least concern' species, $R_F = 0.3$ for 'near threatened', and $R_F = 0.1$ for all threatened species. They further suggested that a value of $R_F = 1.0$ may be appropriate for 'least concern' species known to be increasing or stable.

The following levels of fisheries-related mortality were included in the models:

a) As a conservative approach, a 'high' annual loss of 1 950 and 11 500 shy and white-capped albatrosses, respectively, per year were used (Table 2) based on the updated mortality estimate, and adjusted to account for a high cryptic mortality.

b) Lower mortality levels of 940 and 6 780 shy and white-capped albatrosses, respectively, per year, were used to reflect a lower level of cryptic mortality,

c) Mortality levels equal to the Potential Biological Removal (Wade 1998) were calculated for each species using a range of recovery factors as follows:
— $R_F = 1.0$, 1 524 and 8 887 birds/year for shy and white-capped albatross, respectively;
— $R_F = 0.5$, 762 and 4 444 birds/year for shy and white-capped albatross, respectively;
— $R_F = 0.2$, 305 and 1 777 birds/year for shy and white-capped albatross, respectively; and
— $R_F = 0.1$, 152 and 889 birds/year for shy and white-capped albatross, respectively;

Table 2: Seabird bycatch assessments for fisheries within the range of shy and white-capped albatrosses. Estimates from Chapter 1 are shown, together with revised estimates (where available), adjusted for cryptic mortality. Cryptic mortality multipliers used when this was not accounted for with revised estimates are 2.08 for longline fisheries, and 3.0 (low) and 8.23 (high) for trawl fisheries, respectively (see text).

Data sources: [1]Petersen et al. (2009); [2]Maree et al. (2014); [3]BirdLife 2013a; [4]BirdLife 2013b; [5]Phillips et al. 2010; [6]Richard & Abraham (2014); [7]Jimenez et al. (2012, 2015).

Fishery	'Shy type' albatrosses killed annually		**Low** Cryptic mortality - adjusted estimate for 'Shy type albatrosses		Estimated potential annual mortality by species		**High** Cryptic mortality - adjusted estimate for 'Shy type albatrosses		Estimated potential annual mortality by species	
	2007 estimate	revised estimate	multiplier	estimate	Shy	White-capped	multiplier	estimate	Shy	White-capped
South African PLF	500-600	600 [1]	2.08	1,248	62	1,186	2.08	1,248	62	1,186
South African DTF	5 000	29 [2]	incl.	29	1	28	incl.	29	1	28
Namibian PLF	190	0 [3]	2.08	0	0	0	2.08	0	0	0
Namibian DTF	960	0 [4]	3.0	0	0	0	8.23	0	0	0
Asian DWLF	1 300	1,300	2.08	2,704	135	2,569	2.08	2,704	135	2,569
Australian tuna fisheries	10	10	2.08	21	7	14	2.08	21	7	14
Australian SESSF longline	12	12	2.08	25	9	16	2.08	25	9	16
Australian SESSF trawl	Not assessed	603 [5]	3.0	1,809	633	1,176	8.23	4,963	1,737	3,226
NZ trawl	482-543	504 [6]	3.0	1,513	76	1,513	incl.	4,150	0	4,150
NZ longline	41-77	125 [6]	2.08	259	13	259	incl.	259	0	259
Uruguay pelagic longline		10 [7]	2.08	21	0	21	2.08	21	0	21
Totals	8,496 - 8,693	3,193		7,628	**937**	**6,780**		13,419	**1,952**	**11,467**

Results

The Base Model (with no fisheries-based mortality applied) showed shy albatross population growth of r = 0.006 and a mean final population size of 72 670 individuals (SD = 10 702) after 30 years of modelling (Model 1, Table 3). For white-capped albatross, the base model showed population growth of r = 0.003 and a mean final population size of 473 319 individuals (SD = 72 908) over the same time period (Model 8, Table 4).

Four of the six bycatch scenarios modelled for shy albatross all showed negative population growth (Table 3), with a probability of quasi-extinction of 0.99 and 0.79 for high level fishing mortality and the high PBR, respectively (Table 3, Models 2 and 4). Mean final population size for these models was 13 307 and 24 546, respectively. For scenarios with lower level fisheries-based mortality (Models 3, Table 3) and that using a PBR estimate based on a recovery factor of 0.5 (Model 5, Table 3), fewer populations (probabilities of 0.07 and 0.01, respectively) approached the quasi-extinction level of 30 000 individuals under the 2000 simulations run for each, but mean population size was reduced to 42 976 and 48 419, respectively. Only models with PBR estimates based on conservative Recovery Factors of 0.2 and 0.1 (Table 3, Models 6 and 7) showed positive population growth, with no populations reaching quasi-extinction, and mean final population size of 63 038 and 68 344 after 30 years, respectively.

For the white-capped albatross the results were similar (Table 4). Bycatch scenarios modelled with both high and low estimated bycatch, and those with estimated PBRs based on a recovery factor or 1.0 or 0.5 all showed negative population growth (Table 4, Models 9, 10, 11 and 12). When a high level of fisheries based mortality was modelled, the probability of the population halving in 30 years was 0.87 (Table 4, Model 9). This was drastically reduced to 0.07 under a lower estimated rate of fishing mortality (Table 4, Model 10). The highest PBR value saw a negative population growth rate (r = -0.019) and a reduction in final population size of 45% over 30 years to 231 349 (Table 4, Model 11). Again, as for shy albatross, only models with PBR estimates based on conservative Recovery Factors of 0.2 and 0.1 (Table 4, Models 13 and 14) showed positive population growth, with no populations reaching quasi-extinction.

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Table 3: Model predictions for the shy albatross population showing mean stochastic population growth rate (Stochastic *r*), probability of quasi-extinction (population halving), and mean final population size (N) and standard deviation (SD) after 30 years. Each model was run for a 30 year period with 2000 simulations. Initial population size was 60 000 individuals.

Model	Description	Model explanation	Mean population change	Probability of quasi extinction (population halving)	Mean final population size	
			r		N	SD
1	Base Model	No bycatch mortality	0.006	0	72 670	10 702
2	Base Model+ fishing mortality 1 950/year	High level bycatch mortality	-0.033	0.99	13 307	5 293
3	Base Model + fishing mortality 940/year	Lower level fisheries-based mortality	-0.012	0.07	42 976	8 933
4	Base Model + fishing mortality equal to calculated PBR 1 524/year	PBR estimate based on F_R = 1.0	-0.03	0.79	24 546	7 528
5	Base Model + fishing mortality equal to calculated PBR 762/year	PBR estimate based on F_R = 0.5	-0.008	0.01	48 419	9 618
6	Base Model + fishing mortality equal to calculated PBR 305/year	PBR estimate based on F_R = 0.2	0.001	0	63 038	10428
7	Base Model + fishing mortality equal to calculated PBR 762/year	PBR estimate based on F_R = 0.1	0.004	0	68 344	10 663

Table 4: Model predictions for the white-capped albatross population showing mean stochastic population growth rate (Stochastic *r*), probability of quasi-extinction (population halving), and mean final population size (N) and standard deviation (SD) after 30 years. Initial population size was 424 000 individuals. Each model was run for a 30 year period with 2000 simulations.

Model	Description	Model explanation	Mean population change	Probability of quasi extinction (population halving)	Mean final population size	
			r		N	SD
8	Base Model	No bycatch mortality	0.003	0	473 319	72 908
9	Base Model+ fishing mortality 11 500/year	High level bycatch mortality	-0.027	0.87	153 806	51 120
10	Base Model + fishing mortality 6 780/year	Lower level fisheries-based mortality	-0.012	0.07	297 720	62 819
11	Base Model + fishing mortality equal to calculated PBR 8 887/year	PBR estimate based on F_R = 1.0	-0.019	0.39	231 349	57 853
12	Base Model + fishing mortality equal to calculated PBR 4 444/year	PBR estimate based on F_R = 0.5	-0.005	0.03	373 340	68 375
13	Base Model + fishing mortality equal to calculated PBR 1 777/year	PBR estimate based on F_R = 0.2	0.002	0	456 923	74 084
14	Base Model + fishing mortality equal to calculated PBR 889/year	PBR estimate based on F_R = 0.1	0.004	0	486 174	75 499

Discussion

PVA is useful for guiding conservation management and research by identifying the key demographic parameters and impacts that may be affecting the survival of a species. This chapter has assessed the key demographic factors driving trends in the population of two closely-related albatrosses that are impacted by a number of commercial fisheries operating across the southern hemisphere. In particularly, the modelling ascertained if the currently measured level of incidental mortality from all fisheries is sustainable.

The mean population growth of both shy and white-capped albatrosses under even the best modelled scenarios (i.e. no fisheries-related mortality, Table 3, Model 1 and Table 4, Model 8) was very low ($r = 0.006$ and 0.003, respectively). This indicates that, even in the absence of incidental fisheries-related mortality the populations of both species have the potential to decline under adverse environmental conditions.

Using the high level fisheries-based mortality estimates in the models led to rapid population declines in both shy and white-capped albatrosses, with both populations showing high probability of halving within 30 years ($p = 0.99$ and 0.87, respectively). However, population declines of this level have not been observed for either white-capped (Chapter 2) or shy albatrosses (Alderman et al. 2011) over the last 10 and 20 years, respectively, casting doubt on the veracity of the observed bycatch estimates. The cryptic multipliers used to derive the total bycatch estimates seem most likely responsible for these overestimates. Nevertheless, applying a lower cryptic multiplier of 3 to the estimates of observed bycatch in trawl fisheries still led to negative population growth and reductions in final population size. For white-capped albatross, predicted mean final population size under this scenario was reduced by 30% over 30 years to 297 720; for shy albatross mean final population size was reduced by 28 % to 42 976 over a similar time span.

Again, these predictions do not reflect observed values (Alderman et al. 2011; Chapter 2) suggesting they are still overestimates.

Fisheries observer data are generally poor for most fisheries (see Chapter 1), but are usually the available data for informing insights into the impacts of fishing on non-target species. Agnew (2001) suggested that the level of observer coverage needed to accurately estimate bycatch levels in longline fisheries was 20% of all hooks set. Similar principles are likely to apply to fishing effort for other gear types (Chapter 1). This level of observer coverage is rarely achieved and when coverage is low or unrepresentative, extrapolations used to estimate levels of annual capture are potentially inaccurate and misleading (Chapter 1). The fisheries data used as the basis for estimating observable captures in both Chapter 1 and in this Chapter were largely drawn from samples where observer coverage was less than 20%, therefore placing too much emphasis on the derived estimates is probably inappropriate. Notable exceptions to this are the New Zealand deepwater trawl fisheries (Richard and Abraham 2014).

Similarly, estimators for assessing cryptic mortality are also of low reliability, and there have been few studies that have accurately quantified the extent of this problem with any real confidence (Watkins et al. 2008; Abraham 2010; Brothers et al. 2010; Parker et al. 2013, Richard and Abraham 2014). The approach of Richard and Abraham (2014) in New Zealand in developed cryptic mortality multipliers that are gear and species specific is appropriate, but the relevant values derived for white-capped albatrosses (2.08 and 8.23 for longline and trawl gears, respectively) seem high based on the models presented here, and therefore should probably be reviewed. I concur with the views of Richard and Abraham (2014) that poor knowledge of cryptic mortality restricts understanding of the impacts of fisheries on seabird populations, particularly for trawl fisheries. Widespread use of 'corpse catchers', a warp attachment device developed to improve retention, and hence detection, of seabirds drowned on trawl warps, would go some way toward estimating the number of seabirds killed but not hauled aboard (and therefore not accounted for in estimates of observed bycatch estimates in trawl fisheries – see Parker et al. 2013).

Richard and Abraham (2014) used a modified Potential Biological Removal approach in New Zealand to calculate a risk ratio in a risk assessment framework for a range of seabirds known to interact with fisheries. It is therefore of concern that the calculated PBR for white-capped albatross of 4 040 (2 590-6 340, 95% C.I) is not sustainable as evidenced by the results of Model 12 (Table 4). Rather than ensuring positive population growth, under this scenario population growth rate was negative (r = -0.005) and population size was reduced by 12% over 30 years to 377 340. The use of a recovery factor (F_R) of 1 when calculating the PBR level, as undertaken by Richard and Abraham (2014), does not accord with the recommended guidelines of Dillingham and Fletcher (2008), and a value of $F_R = 0.3$ or lower would be more appropriate, given that the species is considered to be Vulnerable under the IUCN RedList Criteria (Garnett et al. 2011).

While uncertainty remains around the total level of bycatch that both shy and white-capped albatrosses are experiencing, it is likely that total bycatch is higher than may be sustainable in the longer term, and may have been so for many years. Historically, shy albatross were known to be impacted by longline fishing in the 1980s and 1990s by Japanese longline fishing effort in southern Australia (Gales et al. 1998), and white-capped albatrosses by squid-trawling around the Auckland Islands (Bartle 1991). Fishing effort is dynamic and changes regularly over time for economic and stock-related reasons, and understanding of the impacts of incidental mortality on albatross populations requires continual assessment and refinement of estimation techniques, coupled with population monitoring programmes. Notwithstanding the need for better estimates of total mortality, fisheries managers should look to reduce mortality to below conservative PBR values. My modelling would suggest that application of a PBR with $F_R = 0.1$ or $F_R = 0.2$ would be appropriate.

FAO guidelines recommend assessing and minimizing bycatch, irrespective of population-level impacts (FAO 1995). Application of PBRs to manage bycatch sustainably can be effective in maintaining populations at prescribed levels, but a focus also needs to be maintained on the development and implementation of measures to mitigate the bycatch of albatrosses and other seabirds. Considerable work has been done on development of mitigation measures for longline and trawl gear in recent years (reviewed in Løkkeborg 2011; ACAP 2013 a, b, c) but widespread uptake of appropriate measures has not been achieved, despite education programmes and the potential for some forms of mitigation to provide incentives through improved catches. Finding solutions to bycatch problems requires a mix of legislative and political measures to encourage the engagement of industry and provide incentives for action together with robust and evidence-based science to define problems and develop technological solutions. While some fishing companies and individuals have shown a willingness to work toward finding solutions, significant bycatch-mitigation progress is unlikely to be made across all fisheries until fishers are required, by a combination of legislation, compliance and enforcement activities, to address the issues that lead to incidental mortality in their fisheries.

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Chapter 4

Efficacy of the 'Smart Tuna Hook' in reducing bycatch of seabirds in the South African Pelagic Longline Fishery

Abstract

While considerable progress has been made in mitigating bycatch in demersal longline fisheries proven and accepted seabird avoidance measures in pelagic fisheries require substantial improvement. I report on an at-sea experiment to test the efficacy of a mitigation method known as the Smart Tuna Hook (STH). This method uses a modified tuna longline hook which accepts a specially designed shield that disarms the hook once it has been baited, preventing ingestion and making it difficult for any seabird to be hooked. The shield is released within 15 minutes after the hook has been immersed in salt water, allowing fish to be caught after the baited hook has passed beyond the normal diving and feeding depths of most seabirds. After release from the hook the shield sinks to the seafloor where it corrodes within 12 months, leaving no pollution or toxic residue. The byproduct is iron oxide and carbon.

Our experimental work was conducted on pelagic longline vessels targeting tuna and swordfish out of Cape Town, South Africa during the Austral spring of 2014. Seabird bycatch was high and a total of 13 birds were caught across the three trips. Eleven of these birds were caught on the control treatments and 2 birds on the STH treatments. The use of the Smart Tuna Hook led to a reduction in the bycatch of seabirds of between 81.8% – 91.4% in one of the highest-risk fisheries to seabirds in the world. Importantly, there was no detectable difference between setting methods in the catch rates of commercially valuable species, indicating no detectable detrimental effect on fish catch for any species. In a fishery where the bycatch rate of seabirds exceeded 1 bird/1000 hooks (this study), and where the capture of more than 25 birds by a vessel each season leads to a suspension of fishing activity for that vessel, the Smart Tuna Hook clearly provided a significant deterrent to seabirds attacking baits, and offers a feasible option for pelagic fishers to significantly reduce the level of interactions with seabirds and hence remain active in the fishery.

Introduction

Each year many thousands of seabirds are accidentally killed on longline hooks when birds, attracted to fishing vessels by discards and baits, ingest baited hooks and subsequently drown (Anderson et al. 2011; Baker et al. 2002). While most mortality occurs directly when birds are caught during line-setting and, less commonly, hauling, seabirds may also die after they are released with critical injuries, or through ingestion of fishing hooks when birds eat discarded baits and fish heads containing hooks.

The level of longline-related mortality is such that longline fishing has been identified as a major threat affecting many seabirds (Anderson et al. 2011; Gales 1998; Baker and Wise 2005), causing widespread declines in populations throughout the world (Alexander et al. 1997; Birdlife International 1995; Croxall 1998; Delord et al. 2005; Gales 1998; Poncet et al. 2006; Tuck et al. 2001). Most of the larger albatrosses and petrels that breed and forage within the southern hemisphere are threatened by longline fishing (Gales 1998).

A range of mitigation measures for reducing the incidental catch of seabirds in longline fisheries have been developed (Brothers et al. 1999; Dietrich et al. 2004; Bull 2007; Lokkeborg 2008, 2011) that can be employed according to circumstance. While considerable progress has been made in mitigating bycatch in demersal longline fisheries (e.g. Moreno et al. 2007), principally through the development of effective bird scaring lines (Melvin 2003; Melvin et al. 2004), Integrated Weight Line in autoline systems (Robertson et al. 2006), night setting and seasonal closures (SC-CAMLR 2005), proven and accepted seabird avoidance measures in pelagic fisheries require substantial improvement. In 2007, ACAP's Seabird By-catch Working Group reviewed available research on seabird by-catch mitigation measures for pelagic longline fishing (ACAP Seabird Bycatch Working Group 2007; also see Melvin and Baker 2006). They concluded that night setting is currently the only mitigation measure proven to be widely effective with pelagic longline gear, but its widespread adoption is constrained because it is considered to reduce operational efficiency when targeting some pelagic fish species.

In an attempt to address this situation an Australian company, AHI Enterprises, has been working on the development of a mitigation device that will significantly reduce seabird bycatch in pelagic longline fisheries in particular, but will also have utility in other longline gear types. Resulting from this work is a device that is known as the Smart Tuna Hook System (STH - Figure 1). The system uses a modified tuna long-line hook, circle or Japanese style, which accepts a specially designed shield that disarms the hook once it has been baited. The steel shield, once attached to the baited hook, creates a large 3 dimensional barrier encompassing the hook's point and barb, which prevents ingestion and making it difficult for any seabird or turtle to be hooked, internally or externally. The shield is easily and quickly snapped and held onto the baited hook by a clip that has a corrodible alloy link. The link causes the shield to be released within 15 minutes after the hook has been immersed in salt water, allowing fish to be caught after the baited hook has passed beyond the normal diving and feeding depths of most seabirds. After release from the hook the shield sinks to the seafloor where it corrodes within 12 months, leaving no pollution or toxic residue. The byproduct is iron oxide and carbon.

The smart tuna hook works to reduce seabird bycatch in two ways:

1. It adds weight (38 g) to each branchline directly at the hook, thus increasing sink rate and reducing the availability of baited hooks to seabirds. Tests in still, fresh water with an artificial bait on hooks showed the shield placed at the hook improved the sink rate to 4 m depth by 35 % compared to conventionally weighted gear (60 g weight 3.5 m from the hook; 0.60 m/s versus 0.39 m/s; B.Baker and G.Robertson, unpublished). Unlike weighted swivels or fixed weights, however, the weight is not present at the time of hauling because the shield has been released during the soak time. This addresses the safety concerns of many fishers that

weights applied to gear can potentially injure crew members in 'bite-off' or break off situations during gear retrieval (Sullivan et al. 2012).

2. The shield protects the hook from ensnaring seabirds and being ingested in the event that diving birds manage to seize a baited hook during deployment.

Figure 1. The Smart Tuna Hook system showing a tuna hook fitted with a shield (left panel) and a baited hook and shield prior to deployment.

The Smart Tuna Hook presents other potential operational advantages to fishers that will encourage uptake if the environmental benefits can be demonstrated by experimental work. Firstly, lead loss to the marine environment from lost fishing gear will be reduced, as fishers will not need to use weighted swivels to sink gear. This will benefit fishers through a reduction in costs and potential increase in crew safety. Secondly, bait loss during the setting process will be significantly reduced through minimised seabird attacks following improved hook sink rates and protection of the hook. Bait loss is known to be as high as 15% in pelagic fisheries

(Hans Jusseit, unpublished), and the shield will also act to protect the bait from tearing off the hook when it enters the water after being cast from the vessel.

Successful pilot studies of the STH in the Coral Sea, Australia, have shown the system is operationally effective and had no impact on efficient setting of gear (Jusseit 2010). Limited testing at-sea off Ka ikoura, New Zealand, indicated that use of the Smart Tuna Hook was a significant deterrent to seabirds attacking baits. However, support and feedback from stakeholders indicated a larger trial under experimental conditions was warranted before uptake and the production of the device was put into full commercial production.

In 2014 I conducted an at sea trial/experiment setting Smart Tuna Hooks and shields to demonstrate the efficacy of this measure in reducing seabird bycatch whilst maintaining catch of target species. The experimental work was conducted on pelagic longline vessels targeting tuna and swordfish out of Cape Town, South Africa during the Austral spring of 2014. In this paper I report on this experiment and the effectiveness of the Smart Tuna Hook in mitigating capture of seabirds while maintaining or improving capture of target species.

Methods

General

Field work to test the efficacy of the 'Smart Hook' System was carried out between July-October 2014 off the coast of South Africa, a well-known seabird 'hotspot' that is readily accessible for both seabird observation and research work. Twenty four species of albatrosses and petrels have been recorded foraging in South African waters (Petersen et al. 2009), and bycatch rates in South African Pelagic Longline Fisheries are known to be high (Melvin et al. 2013). Baker et al. (2007) assessed this fishery to be a very high impact fishery for seabirds with >1000 seabirds killed in the fishery each year. Fishing effort is divided between domestic vessels that primarily target broadbill swordfish (*Xiphias gladius*), and foreign (Japanese, Korean, Taiwanese) fishing vessels that usually set their gear deeper and target albacore (*Thunnus alalunga*), yellowfin (*T. albacares*) and bigeye tuna (*T. obesus*) (Cooper and Ryan 2002). Effort is usually concentrated along the edge of the continental shelf although some vessels fish farther offshore in the Atlantic and Indian Oceans, outside the South African EEZ (Cooper and Ryan 2002). More detailed descriptions of the fishery are provided in Baker et al. (2007) and Melvin et al. (2013).

The experiment involved a direct comparison of the Smart Tuna Hook and conventional pelagic hooks in pelagic fishing operations. The experiment primarily focussed on the efficacy of mitigating seabird by-catch using the Smart Hook in temperate waters, but data were also recorded on catch of target species and interactions with other non-target species.

Fishing vessels and gear

The experiment was conducted on two fishing vessels, the *FV Seawin Emerald* and *FV Seawin Diamond*. Both these vessels are 25-m steel hulled longliners operating out of Cape Town, South Africa. Gear configuration and operational procedures during fishing were similar for both vessels. They set a 3.2mm monofilament mainline without a line shooter. The mainline was suspended on floats on 17m long droppers. Branch lines were made of 1.8mm monofilament nylon, 13.4 m long and were fitted with a mix of #14/0 J hooks and circle hooks. An 80 g weighted swivel was placed 3.2 m from each hook. Baited hooks were deployed to the outer edge of vessel wake on both sides of the vessel. The bait was whole Argentine short-fin squid (*Illex argentinus*). All baits were dead. Branch lines with squid bait were always accompanied by a green light stick placed 2 m from hooks. A typical set on each vessel involved deploying 900–1500 hooks at 8 knots vessel speed with 4 branch lines between floats and branch lines 50 m apart. Branch lines were set from gear bins every 8 s off both sides of the vessel. Radio beacons were deployed at 130 branch line intervals. The main species targeted in the experiments were broad-bill swordfish (*Xiphias gladius*), yellow-fin tuna and big-eye tuna. Time-of-day of line setting varied with moon phase and operational issues, but in general commenced at nautical dusk as swordfish was the main target species. All sets of experimental gear occurred during the night.

This experiment was conducted over three fishing trips and 28 longline sets. The experiment consisted of a comparison of a) a control of conventional surface setting of pelagic long-lines from the stern of a vessel using hooks employing a form of conventional mitigation regulated by the jurisdiction (see below) within which the experiment was undertaken but without the 'Smart Hook' shield (the 'conventional method' or control), and b) the same gear as described for the control but with the 'Smart Tuna Hook' shield fitted to each hook. The regulated mitigation regime used in the control consisted of an 80 g weighted swivel placed 3.2 m from each hook. Currently, pelagic long-line vessels worldwide set gear using mainly method 'a'. It was originally intended to place the Smart Tuna Hook shield on hooks fixed to branchlines without weighted swivels, but the crew were unwilling to use unweighted branchlines in experimental work, and the conventional and STH methods therefore differed only in the presence or absence of Smart Tuna Hook shields. The experiment sought to examine the capacity of the two hooking methods to deter seabird species known to readily interact with pelagic long-line fishing gear, particularly albatrosses, white-chinned petrels and shearwaters. The response variables were the number of seabirds caught and the number of target and non-target fish species caught. Because of concerns of statistical power, work was undertaken using a simple 1 X 2 factorial approach:

Factors	**Treatments**	
1. Bait Type	**Squid**	
2. Hook method	No Smart Hook Shield	Use of smart hook shield

Both treatments were tested during each longline set to eliminate the potentially confounding effect of 'set'. Both treatments were tested in a block (experimental set) of c.120-150 hooks for each treatment on the longline, with a length of c.500m of mainline to isolate each treatment in the set and ensure a degree of independence between treatments. The number of hooks set in each block differed in each block and was set to suit the normal operations of the vessel with respect to total hooks set, in order to ensure that evaluation of each treatment was balanced. For example, if the vessel was intending to set 1500 hooks per set, each block for that set was standardised at 150 hooks, so that 10 blocks, or five paired blocks, could potentially be set each day. The setting order of treatments during each set was randomised.

No abundance estimates or other records of seabirds attending the vessel were made as fishing was undertaken during the night.

Statistical models and methods

The response variables modelled for each of control hooks and Smart Tuna Hooks (STHs) in each pair of deployments were (a) the number of seabirds caught (i.e. tallied across the 3 species of black-browed albatross, shy albatross, and white-chinned petrel); (b) number of fish totalled across the 5 main commercial species caught; and (c) the number of fish for each of the 5 main commercial species caught of albacore tuna (ALB), yellowfin tuna (YFT), bigeye tuna (BET), southern bluefin tuna (SBF), and swordfish (SWO). The response variables were modelled using generalized linear mixed models (GLMMs) (Bolker et al. 2009) using a Poisson error distribution and the log of number of hooks within each hook type within each pair as an offset. The linear predictor included just the single factor of hook type (**`Treat_f`**) (i.e. control vs STH). For the GLMM random effects of pair within trip (**`Pair_f`**) and trip number (**`TripNr_f`**) were included in the conditional linear predictor. The GLMM was fitted using `glmer` (Bates et al. 2014) function in the `lme4` library in R (R Core Team 2013) where this function evaluates the marginal likelihood by approximation of the integral of the conditional likelihood across the assumed Gaussian distributions for random effects. Robertson et al. (2006) modelled seabird bycatch in paired deployments of two line types in the demersal ling fishery on the west coast of New Zealand using a similar Poisson GLMM except that Penalized Quasi-likelihood Estimation (`PQL`) (Schall 1991; Breslow and Clayton 1993) was used for estimation. Marginal likelihood though more numerically intensive has superior estimation properties to PQL in some situations but is always at least as good as PQL. Irrespective of this, the response variable of total seabird bycatch had a large proportion of zeros for with only 2 out of 63 STH pair deployments being non-zero and only 9 out of 63 control pair deployments being non-zero. However, the Poisson distribution, even with a very low expected value for rate of the order of 0.19 per 150 hooks (see results for controls in Supplementary Information `MCMCglmm` output) (where the average number of hooks per STH or control deployment had a range of 75 to 150 with median of 150 and mean of 139.3) has difficulty modelling such a large proportion of zeros. For this reason a zero-inflated Poisson (ZIP) distribution model was applied which is not a standard GLM and cannot be fitted using `glmer`. Further, the response variables of number of fish caught for each commercial species also contained a high proportion of zeros of the order of 50%.

So in order to incorporate random effects in the ZIP model, the `MCMCglmm` function in the `MCMCglmm` R-library (Hadfield 2010) that applies Markov Chain Monte Carlo sampling (Gilks et al. 1996) was employed for these response variables. The offset of log of number of hooks was included in `MCMCglmm` for the "trait" of the Poisson latent component (Hadfield 2014) by setting a regression parameter associated with the variable log of number of hooks with prior Gaussian distribution with mu of 1.0 and variance of 1e-9. Other parameters were given diffuse priors (Hadfield 2010).

For both `glmer` and `MCMCglmm` the output parameters of rate of seabird bycatch per hook and fish catch per hook for each of control and STH deployments were parameterized as a single parameter of the percent reduction in bycatch rate (for seabirds) and catch rate (for fish) of the STH relative to the control deployments ( ) (i.e. this definition does not preclude an increase in catch rate in which case this parameter would have a negative estimate). Using `glmer` output (see Supplementary Information) and using the parameter defining catch rate on the log scale for STH minus that for the control, of $\hat{\tau}$ then $\hat{R} = 100\left[1 - \exp\{\hat{\tau}\}\right]$ (See Supplementary Information for glmer output), the fixed effect parameter "`Treat-fCap`" is equivalent to $\hat{\tau}$. Approximate 95% confidence intervals were obtained for $\hat{R}$ using `glmer` with lower limit of $100\ (1 - \exp\left\{\hat{\tau} + 2se\left(\hat{\tau}\right)\right\}$ and upper limit of $100\ (1 - \exp\left\{\hat{\tau} - 2se\left(\hat{\tau}\right)\right\}$. In contrast using the `MCMCglmm` output (see Supplementary Information), MCMC sample values for (by)catch rate per hook for each of STH (α_{STH}) and control (α_{con}) deployments was obtained. The estimate of R was taken as the median of the sample of values with ith sample value of $R_i = 100\left(\hat{\alpha}_{con,i} - \hat{\alpha}_{STH,i}\right) / \hat{\alpha}_{con,i}$. Confidence bounds with 95% support were obtained as 2.5% and 97.5% quantiles of the set of sample values of R. Two MCMC estimations were carried out for the total seabird bycatch, one with 1 000 sample size using a burn-in sample of 200 000, total sample of 700 000, and thinning rate of 1 in 500. The second MCMC with the thinning rate dropped to 1 in 250 to give a final sample of 2 000. The fish catch for each commercial fish species use the first of these two MCMC sampling strategies since the samples were better behaved presumably because of a less extreme number of excess zeros.

Results

General

Gear loss was minimal for most sets, but significant sections of the treatments C2Cap, C2 Con, H1Cap, H2Cap, and H3Con were lost when multiple line breaks lead to complete loss of gear.

Seabird bycatch was high and a total of 13 birds were caught across the three trips. Eleven of these birds were caught on the control treatment and 2 birds on the STH treatment. All birds caught had either been hooked through the beak or mouth. The two birds caught on the STH treatments had been hooked through the throat just below the beak. The one White-chinned Petrel that was caught on the STH treatment came up tangled in line and was effectively attached to the mainline, indicating a fault during the setting process, potentially because the STH was deployed with a weighted swivel on the branchline as well as the weight of the STH at the hook, and the weights fell either side of the mainline on casting, hence encouraging a tangle.

Seabird catch

Two sets of analysis were carried out for total seabird bycatch. The first set used all the data. The second set dropped the pair 3J1, both control and STH values, since a single WCP was caught on the STH but this was considered potentially unrepresentative due to a faulty deployment of the STH that caught the bird (see discussion). The Trip random effect (**`TripNr_f`**) was estimated to have zero variance by `glmer` so subsequently the only random effect incorporated in `glmer` and `MCMCglmm` was (**`Pair_f`**).

Table 1 gives the results of the fit of the Poisson GLMM and ZIP models in terms of the estimates of R and its approximate 95% confidence bounds for total seabird bycatch. Seabird bycatch rate was estimated to range from 0.647 — 1.411 birds/1000 hooks for the control hooks, and 0.059 — 0.247 birds/1000 hooks for STH (Table 1). The estimated average reduction in bycatch when using the STH was significantly lower when compared with the control, and ranged from 82-91% reduction, depending on the model used and the dataset fitted (Table 1). Dropping pair 3j1 from the analysis increased the average reduction slightly, but dramatically increased the lower confidence bound by around 30 percentage points (Table 1).

It should be noted that due to the very high proportion of zeros in the seabird bycatch data, less confidence should be placed in the GLMM results that used a Poisson error distribution compared to the ZIP model combined with MCMC sampling in which this feature of the data is more realistically modelled.

The Supplementary Information gives `glmer` and `MCMCglmm` function calls and associated R commands as well as selected outputs from both functions for the full dataset and 1,000 final MCMC samples.

Table 1. Percent reduction (R) in rate of seabird bycatch for Smart Tuna Hooks (STH) compared to control hook deployments, and bycatch rate for control hooks and STH estimated using each of Poisson GLMM, and zero-inflated Poisson MCMC sampling.

Estimation Model	**Dataset**	**Percent Reduction (R)**	**Confidence Limits for R (~95% level)**	**Bycatch Rate (seabirds per 1000 hooks) (~95% Confidence Limits)**	
				Control	**Smart Tuna**
GLMM	All 63 pairs	81.82	15.43, 96.09	0.730 (0.211, 2.531)	0.133 (0.223, 0.790)
	Drop 3J1 pair	90.91	26.58, 98.87	0.647 (0.163, 2.561)	0.059 (0.006, 0.617)
MCMC (1000 samples)	All 63 pairs	83.47	36.45, 97.60	1.241 (0.668, 2.203)	0.198 (0.029, 0.697)
	Drop 3J1 pair	91.39	72.50, 98.61	1.304 (0.648, 2.348)	0.110 (0.018, 0.460)
MCMC (2000 samples)	All 63 pairs	82.57	20.91, 95.36	1.411 (0.598, 2.338)	0.247 (0.062, 0.844)
	Drop 3J1 pair	85.76	49.70, 98.19	1.310 (0.672, 2.101)	0.181 (0.027, 0.542)

Table 2. Percent reduction (positive) or increase (negative) in rate of commercial and non-commercial species catch for Smart Tuna Hooks (STH) compared to control hook deployments estimated using Poisson GLMM, and zero-inflated Poisson MCMC sampling and catch numbers by hook type

Response Variable	Method	Percent Reduction (+) or Increase (-)	Confidence Limits (~95% level)	Total Number Caught Control	Total Number Caught STH	Number of zero catches (out of 126 sets within pairs)
Total all commercial species	**GLMM**	**6.95**	**-6.45, 18.67**	**446**	**415**	**12**
albacore tuna	MCMC	31.60	-24.63, 61.01	124	97	60
yellowfin tuna	MCMC	11.04	-58.26, 49.22	143	142	58
bigeye tuna	MCMC	-0.57	-88.56, 47.67	99	93	65
southern bluefin tuna	MCMC	-4.28	-75.18, 39.43	23	25	88
swordfish	MCMC	-10.62	-100.13, 31.25	57	58	64
Total all non-commercial species	**GLMM**	**-2.02**	**-9.58, 5.02**	**1 536**	**1 567**	**4**
Atlantic pomfret (POA)	MCMC	26.38	-1537.38, 98.65	14	45	119
short-finned mako shark	MCMC	9.10	-54.86, 42.00	112	106	46
blue shark	MCMC	1.31	-33.48, 27.45	1 410	1 416	5

Fish catch

Table 2 gives the results of the fit of the Poisson GLMM and ZIP models in terms of the estimates of and its approximate 95% confidence bounds for total commercial fish catch and individual-species of commercial and non-commercial fish catch. There was no detectable difference between setting methods in the catch rates of swordfish, yellow-fin tuna, big-eye tuna, southern Bluefin tuna, albacore tuna and other commercially valuable species (Table 2). A zero value for the percent reduction was always well within the 95% confidence bounds, indicating no detectable detrimental effect on fish catch for any species.

Only with albacore was there any indication that there could be a difference in catch between the control and the Smart Tuna Hook, but the confidence levels for the estimated difference are quite broad and include zero, meaning that the difference could be due to chance alone.

The Supplementary Information gives `glmer` function call and associated R commands as well as selected outputs for total commercial fish catch and output and graphs of MCMC samples for yellowfin tuna as a typical example of all five commercial species.

Discussion

I was able to demonstrate that the use of the Smart Tuna Hook led to a reduction in the bycatch of seabirds of between 81.8% – 91.4% in one of the highest-risk fisheries to seabirds in the world (Anderson et al. 2011; Petersen et al. 2009). In a fishery where the bycatch rate of seabirds exceeded 1 bird/1000 hooks (this study), and where the capture of more than 25 birds by a vessel each season leads to a suspension of fishing activity for that vessel, the Smart Tuna Hook offers a feasible option for pelagic fishers to significantly reduce the level of interactions with seabirds and hence remain active in the fishery. It clearly provided a significant deterrent to seabirds attacking baits.

While some forms of seabird bycatch mitigation are thought by fishers to impact the catch of commercial target species, in our study there was no detectable difference between setting methods in the catch rates of swordfish, yellow-fin tuna, big-eye tuna, southern Bluefin tuna, albacore tuna and other commercially valuable species, indicating no detrimental effect on fish catch for any species. This provides confidence for fishers planning to use the Smart Tuna Hook that in looking to reduce the risk of seabird bycatch their commercial operations will not be negatively impacted. It stands to reason that if seabirds cannot readily access baited hooks because of the protection provided by the STH shield, then bait retention will be improved and the probability of catch of target species enhanced. There is some indication of this from previous work carried out in the Coral Sea, Australia (Jusseitt 2010) but statistical demonstration of this would likely require examination of many thousands of hooks under controlled experimental conditions. However, there would appear to be immediate economic benefits to the South African Pelagic Longline Fishery of using the STH and minimising seabird bycatch, thus greatly reducing the risk of a seasonal closures to individual vessels and subsequent loss of income.

While hook protection systems such as the STH can clearly substantially reduce the incidental capture of seabirds, as demonstrated by this study, they are unlikely to completely eliminate it. Because hooks protected by the STH shield must at some stage become armed to enable fish to be caught, such systems will only protect birds from capture during line setting, and not hauling. The dissolve time for the corrodible link (rivet) that fixes the shield to a baited hook is c. 10-20 minutes (H. Jusseitt unpublished), which should ensure that seabird capture during setting of hooks to fishing depth is minimal. However, I was unable to tell when the two birds caught on the STH treatment were actually captured and a myriad potential causes exist. These include:

(1.) birds were killed on the soak or haul, when shields had been lost and hooks were no longer protected.
(2.) the hook and shield became entangled on the mainline immediately on setting when the bait was at the surface, as observed in one of the captures in this study, and the gear did not sink to fishing depth before release of the shield.
(3.) Baited hooks were taken to the surface by diving species such as white-chinned petrels after the shield had fallen away, where they subsequently become available to other birds (described by Melvin et al. (2013) as 'secondary' attacks). While all gear was set shallow (40 m) during the experimental work, baited hooks can be expected to move or loft within the water column due to currents and upwellings. With snood lengths of 13 m, some hooks could be expected to be fishing at 25 m. depth or so, which is well within the diving capabilities of some shearwater species. Note however that recorded dive depths of white-chinned petrels, one of the most commonly caught species in South Africa, are not known to exceed 16 m (Rollinson et al. 2014).
(4.) A shield failure may have occurred due to a fault in the rivet/pin causing a mechanical failure i.e. the rivet was weakened and detached shortly after being attached to the hook.
(5.) The gap between the clip and shield was distorted, not holding the hook properly.

Mitigation of seabird bycatch in pelagic longline fisheries still presents one of the greatest challenges to gear technologists. Currently no single measure is considered capable of reliably preventing the incidental mortality of seabirds in most pelagic longline fisheries (ACAP 2013). The most effective approach recommended is the use of a combination of weighted branch lines, bird scaring lines and night setting, which is considered to represent best practice (ACAP 2013). Note however, that the use of two of these measures in combination (night setting and weighting of branchlines)was not capable of reducing seabird bycatch to anywhere near the level prescribed as acceptable in South Africa's NPOA-Seabirds (0.05 birds per 1,000 hooks; Department of Environmental Affairs and Tourism, South Africa, 2008). This level was only approached when the Smart Tuna Hook was used in combination with those two measure.

The Smart Tuna Hook provides a safe and efficient solution for commercial fisheries to minimise seabird bycatch in high-risk operations. Uptake by the fishing industry in areas where seabird bycatch is high is strongly recommended, but this will undoubtedly be determined by unit cost and fishing economics. At this stage the STH is not in full commercial production. Shields were produced for the experiment at a unit cost of USD $0.24 per shield, but full commercial production and associated economies of scale are expected to see this cost reduced. As such, the STH should be a viable mitigation measure in high-value fisheries such as those targeting tunas and billfish, but uptake in many of the artisanal longline fisheries where product value is low is unlikely at this stage.

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Supplementary Information

```
> summary(data)
    RecordNr          TripNr          SetNr          Pair_ID    Setting_Method
Vessel         Bait         Hooks          BB_Alb            Shy_Alb
 Min.   :  1.00   Min.   :1.000   Min.   : 1.00   1A1Cap :  1   Control:63     Min.
:1.000   squid:126   Min.   : 75.0   Min.   :0.000000   Min.   :0.00000
 1st Qu.: 32.25   1st Qu.:2.000   1st Qu.:10.00   1A1Con :  1   STH    :63     1st
Qu.:1.000              1st Qu.:129.2   1st Qu.:0.000000   1st Qu.:0.00000
 Median : 63.50   Median :3.000   Median :17.00   1B1Cap :  1                  Median
:2.000              Median :150.0   Median :0.000000   Median :0.00000
 Mean   : 63.50   Mean   :2.286   Mean   :16.29   1B1Con :  1                  Mean
:1.524              Mean   :139.3   Mean   :0.007937   Mean   :0.06349
 3rd Qu.: 94.75   3rd Qu.:3.000   3rd Qu.:23.00   1B2Cap :  1                  3rd
Qu.:2.000              3rd Qu.:150.0   3rd Qu.:0.000000   3rd Qu.:0.00000
 Max.   :126.00   Max.   :3.000   Max.   :28.00   1B2Con :  1                  Max.
:2.000              Max.   :150.0   Max.   :1.000000   Max.   :2.00000

      WCP           Total_birds
Min.   :0.00000   Min.   :0.0000
1st Qu.:0.00000   1st Qu.:0.0000
Median :0.00000   Median :0.0000
Mean   :0.03175   Mean   :0.1032
3rd Qu.:0.00000   3rd Qu.:0.0000
Max.   :1.00000   Max.   :2.0000

> Nd <- dim(data)[1]
> Ndd <- Nd/2
> data$Treat_f <- factor(x=rep(seq(2,1,-1),times=Ndd), levels=c(1,2),
labels=c("Con","Cap"))
> data$Pair_f <- factor(x=rep(c(1:Ndd),each=2), levels=c(1:Ndd))
> data$TripNr_f <- as.factor(data$TripNr)
> data$Lhooks <- log(data$Hooks)
>
> ### total seabird bycatch  ###
>
> ### Poisson GLMM using glmer()
>
> glmer.01 <- glmer(formula=Total_birds ~ 1 + Treat_f + (1|Pair_f),
offset=Lhooks, data=data, family=poisson)
> summary(glmer.01)
Generalized linear mixed model fit by maximum likelihood (Laplace
Approximation) ['glmerMod']
 Family: poisson  ( log )
Formula: Total_birds ~ 1 + Treat_f + (1 | Pair_f)
   Data: data
 Offset: Lhooks

     AIC      BIC   logLik deviance df.resid
    85.0     93.5    -39.5     79.0      123

Scaled residuals:
    Min      1Q  Median      3Q     Max
-0.5020 -0.2982 -0.1680 -0.1247  4.4577

Random effects:
 Groups Name        Variance Std.Dev.
 Pair_f (Intercept) 1.187    1.089
Number of obs: 126, groups:  Pair_f, 63

Fixed effects:
            Estimate Std. Error z value Pr(>|z|)
(Intercept)  -7.2223     0.6215 -11.621   <2e-16 ***
Treat_fCap   -1.7047     0.7686  -2.218   0.0265 *
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

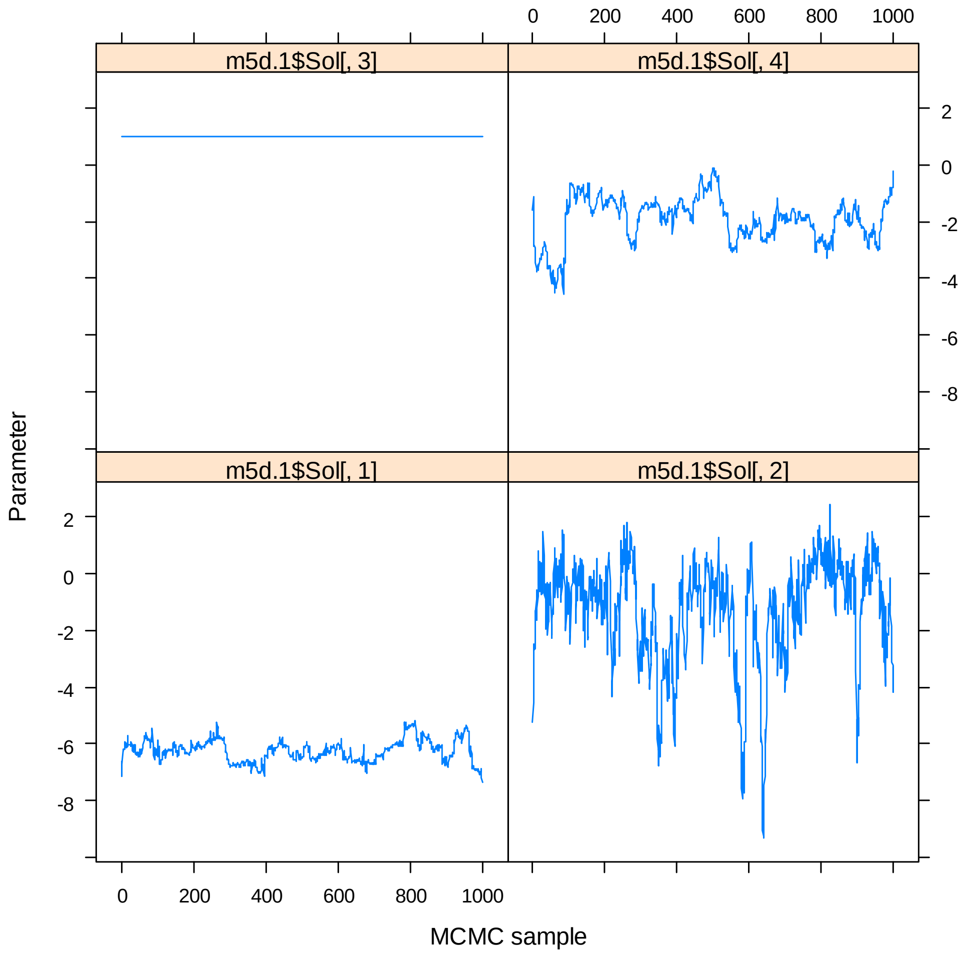
```
Correlation of Fixed Effects:
           (Intr)
Treat_fCap -0.190
>
> exp(fixef(glmer.01)[2])
Treat_fCap
 0.1818178
> vcov(glmer.01)
2 x 2 Matrix of class "dpoMatrix"
            (Intercept)  Treat_fCap
(Intercept)  0.38626842 -0.09087409
Treat_fCap  -0.09087409  0.59068291
> se.teff <- (vcov(glmer.01)[2,2])^0.5
> print(c(fixef(glmer.01)[2],se.teff))
Treat_fCap
 -1.704750   0.768559
> PercReduction <- 100*(1-exp(fixef(glmer.01)[2]))
> sum(fixef(glmer.01))
[1] -8.927003
>
> PercReduction.L1 <- 100*(1-exp(fixef(glmer.01)[2]+2*se.teff))
> PercReduction.L2 <- 100*(1-exp(fixef(glmer.01)[2]-2*se.teff))
>
> print(c(PercReduction,PercReduction.L1,PercReduction.L2))
Treat_fCap Treat_fCap Treat_fCap
  81.81822   15.43350   96.09092


> ### ZIP using MCMCglmm ###


> data$Treat <- as.integer(data$Treat_f %in% "Cap")

>


> prior1 <- list(R = list(V = diag(2), nu = 0.002, fix = 2), B= list (mu =
matrix(c(0,0,1,-1.7),4),V = diag(4)*(10)))

> diag(prior1$B$V)[3]<-1e-9

>

>

> m5d.1 <- MCMCglmm(Total_birds ~ trait - 1 + at.level(trait,1):Lhooks +
at.level(trait,1):Treat, rcov = ~idh(trait):units, data = data,

+      nitt=700000, thin=500, burnin=200000, prior = prior1, family =
"zipoisson", verbose = FALSE)

>

> summary(m5d.1)


 Iterations = 200001:699501

 Thinning interval  = 500

 Sample size  = 1000


 DIC: 81.22326


 R-structure:  ~idh(trait):units
```

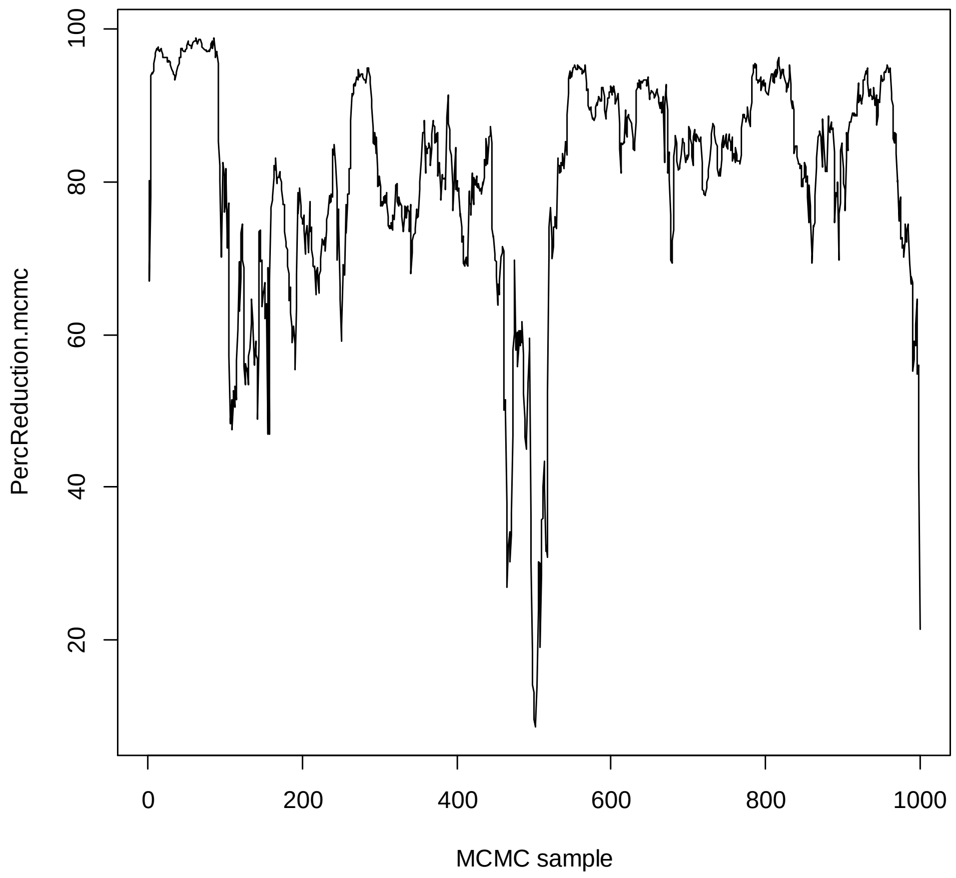
```
                     post.mean  l-95% CI u-95% CI eff.samp
Total_birds.units     0.006042 0.0001553  0.02126    503.8
zi_Total_birds.units  1.000000 1.0000000  1.00000      0.0

 Location effects: Total_birds ~ trait - 1 + at.level(trait, 1):Lhooks + 
at.level(trait, 1):Treat 

                          post.mean l-95% CI u-95% CI eff.samp  pMCMC    
traitTotal_birds            -6.2435  -6.9971  -5.4590    18.11 <0.001 ***
traitzi_Total_birds         -1.2936  -5.2971   1.7077    34.23  0.504    
at.level(trait, 1):Lhooks    1.0000   0.9999   1.0001  1000.00 <0.001 ***
at.level(trait, 1):Treat    -1.9230  -3.6237  -0.3581    11.86 <0.001 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1 
> 
> x <- seq(1,1000)
> xyplot(m5d.1$Sol[, 1]+m5d.1$Sol[, 2]+m5d.1$Sol[, 3] + m5d.1$Sol[, 4] ~ x, 
outer=TRUE, type="l", xlab="MCMC sample")
> savePlot(filename = "Models Pars MCMC.emf", type ="emf")
> 
> xyplot(m5d.1$Sol[, 1]+m5d.1$Sol[, 2]+m5d.1$Sol[, 3] + m5d.1$Sol[, 4] ~ x, 
outer=TRUE, type="l", ylab="Parameter",
+   xlab="MCMC sample")
> savePlot(filename = "Models Pars MCMC.emf", type ="emf")
> 
> summary(m5d.1$Sol[,1])

Iterations = 200001:699501
Thinning interval = 500 
Number of chains = 1 
Sample size per chain = 1000 

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE 
      -6.24345        0.38369        0.01213        0.09016 

2. Quantiles for each variable:

  2.5%    25%    50%    75%  97.5% 
-6.960 -6.535 -6.228 -6.022 -5.381
```

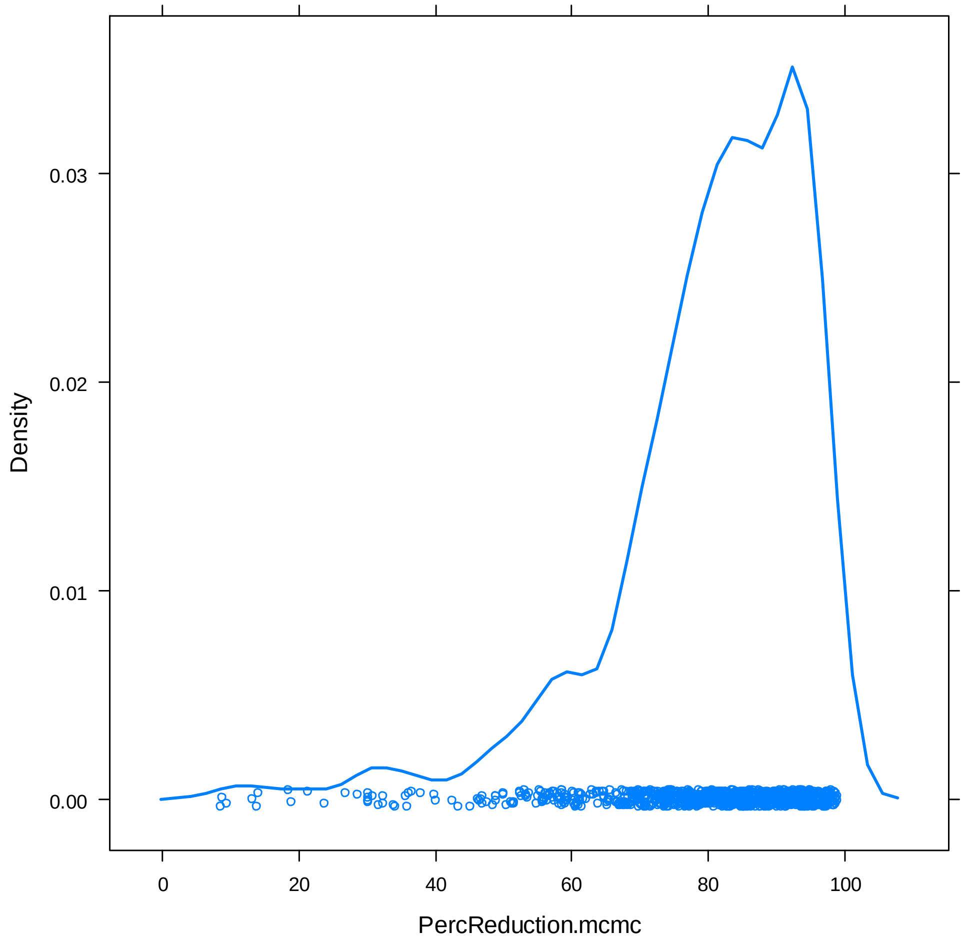
```
>
> c2 <- ((16 * sqrt(3))/(15 * pi))^2
> quantile(plogis(m5d.1$Sol[, 2]/sqrt(1 + c2)))
          0%          25%          50%          75%         100%
0.0003217669 0.1252689677 0.3243001798 0.5006211190 0.8908480698
>
> prob.nz <- plogis(m5d.1$Sol[, 2]/sqrt(1 + c2))
> summary(prob.nz)

Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
      0.328468       0.216151       0.006835       0.032820

2. Quantiles for each variable:

    2.5%      25%      50%      75%    97.5%
0.004825 0.125269 0.324300 0.500621 0.739514

>
> post.prob <- (exp(m5d.1$Sol[,1]))*(1-plogis(m5d.1$Sol[, 2]/sqrt(1 + c2)))
> summary(post.prob)

Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
     1.288e-03      3.989e-04      1.261e-05      3.663e-05
```

```
2. Quantiles for each variable:

     2.5%       25%       50%       75%     97.5%
0.0006676 0.0010007 0.0012407 0.0015302 0.0022033

>1.288e-03*150
[1] 0.1932
>
> plot(y=post.prob, x=seq(1,length(post.prob)), type="l")
>
> post.probCap <- (exp(m5d.1$Sol[,1] + m5d.1$Sol[,4]))*(1-
plogis(m5d.1$Sol[, 2]/sqrt(1 + c2)))
> summary(post.probCap)

Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
     2.353e-04      1.714e-04      5.420e-06      3.734e-05

2. Quantiles for each variable:

     2.5%       25%       50%       75%     97.5%
2.936e-05 1.098e-04 1.982e-04 3.091e-04 6.968e-04

>
> plot(y=post.probCap, x=seq(1,length(post.probCap)), type="l")
>
> PercReduction.mcmc <- 100*(post.prob-post.probCap)/post.prob
>
> plot(y=PercReduction.mcmc, x=seq(1,length(post.probCap)), type="l",
xlab="MCMC sample")
>
> savePlot(filename = "PercReduction.mcmc.emf", type ="emf")
>
> summary(PercReduction.mcmc)
```

```
Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
       80.4782        14.8490         0.4696         3.7051

2. Quantiles for each variable:

 2.5%   25%   50%   75% 97.5%
36.45 74.41 83.47 91.49 97.60

>
> quantile(x=PercReduction.mcmc, probs=c(0.025,0.5,0.975))
    2.5%      50%    97.5%
36.45157 83.47225 97.59782
```


m5d.1$Sol[, 3]
m5d.1$Sol[, 4]
m5d.1$Sol[, 1]
m5d.1$Sol[, 2]
Parameter
MCMC sample
0
200
400
600
800
1000
2
0
-2
-4
-6
-8

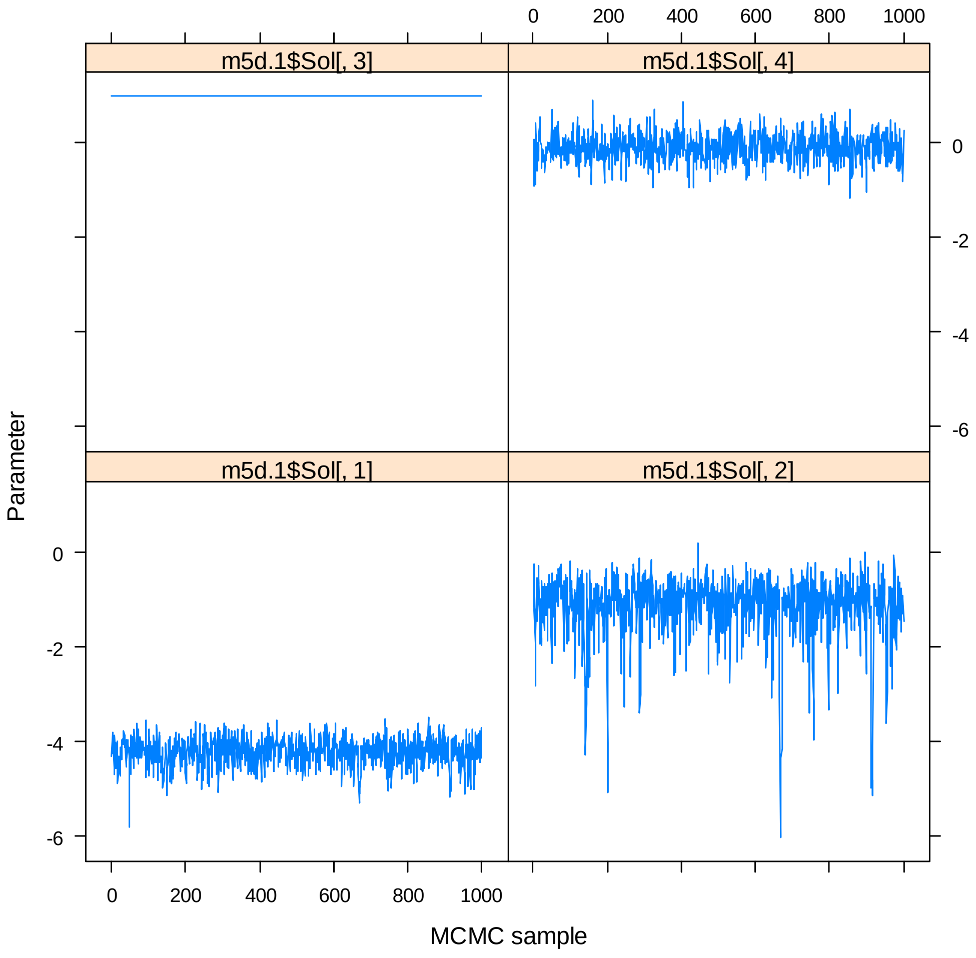

PercReduction.mcmc
100
80
60
40
20
0
200
400
600
800
1000
MCMC sample

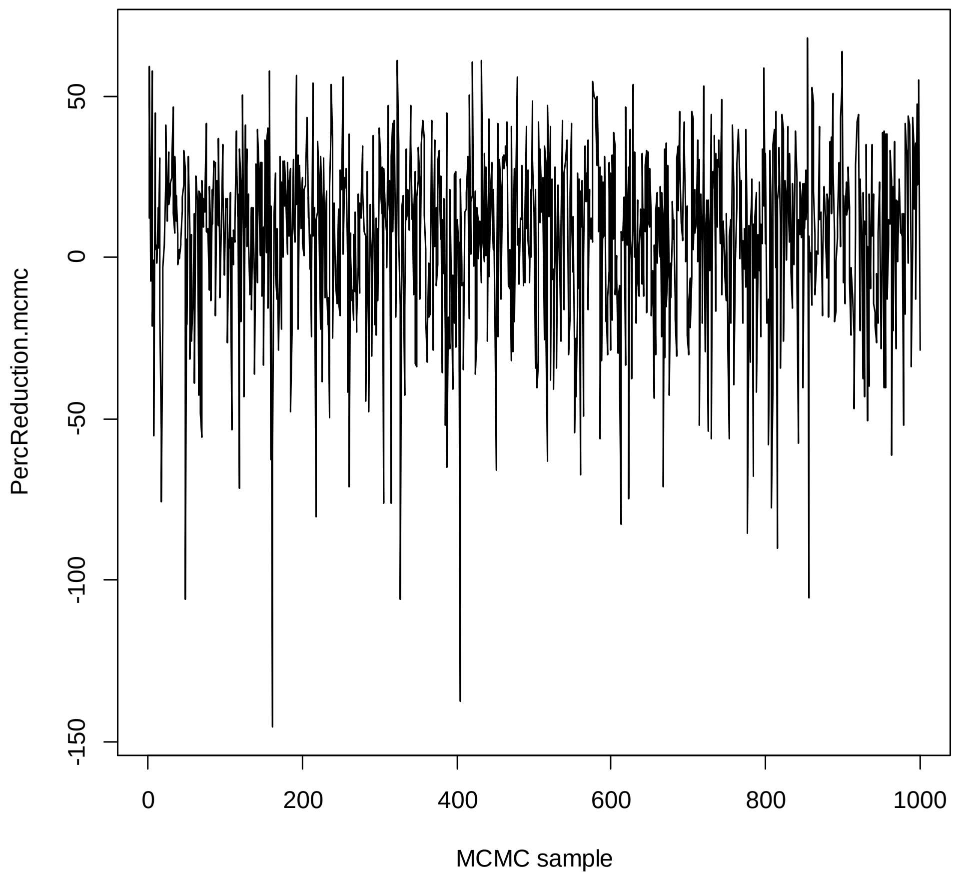


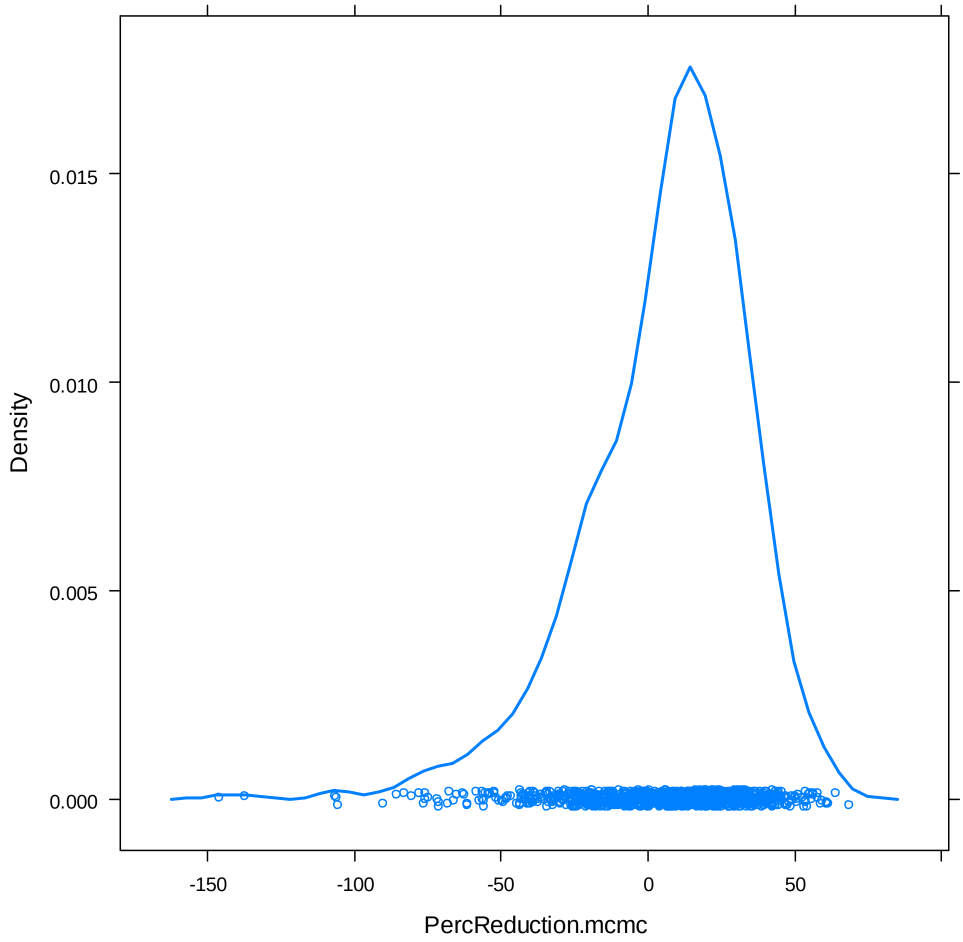

```
>

>  ### Total Commercial Species Catch ###
>
> glmer.01 <- glmer(formula=Total.comm.fish ~ 1 + Treat_f + (1|Pair_f),
offset=Lhooks, data=data, family=poisson)
> summary(glmer.01)
Generalized linear mixed model fit by maximum likelihood (Laplace
Approximation) ['glmerMod']
 Family: poisson  ( log )
Formula: Total.comm.fish ~ 1 + Treat_f + (1 | Pair_f)
   Data: data
 Offset: Lhooks

     AIC      BIC   logLik deviance df.resid
   751.0    759.5   -372.5    745.0      123

Scaled residuals:
    Min      1Q  Median      3Q     Max
-2.0567 -0.9010 -0.1836  0.5766  2.4908

Random effects:
 Groups Name        Variance Std.Dev.
 Pair_f (Intercept) 0.768    0.8764
Number of obs: 126, groups:  Pair_f, 63
```

```
Fixed effects:
            Estimate Std. Error z value Pr(>|z|)
(Intercept) -3.30381    0.12484 -26.465   <2e-16 ***
Treat_fCap  -0.07204    0.06728  -1.071    0.284
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Correlation of Fixed Effects:
           (Intr)
Treat_fCap -0.260
>
> exp(fixef(glmer.01)[2])
Treat_fCap
 0.9304919
>
> se.teff <- (vcov(glmer.01)[2,2])^0.5
>
> print(c(fixef(glmer.01)[2],se.teff))
 Treat_fCap
-0.07204194  0.06728159
>
> PercReduction <- 100*(1-exp(fixef(glmer.01)[2]))
> sum(fixef(glmer.01))
[1] -3.375854
>
> PercReduction.L1 <- 100*(1-exp(fixef(glmer.01)[2]+2*se.teff))
> PercReduction.L2 <- 100*(1-exp(fixef(glmer.01)[2]-2*se.teff))
>
>
> print(c(PercReduction,PercReduction.L1,PercReduction.L2))
Treat_fCap Treat_fCap Treat_fCap
  6.950813  -6.451707  18.665924


> ### yellowfin tuna catch ###
>
[1] "YFT"
[1] 34 # no Control pairs with > 0 catch
[1] 34 # no Cap pairs with > 0 catch
   Min. 1st Qu.  Median    Mean 3rd Qu.    Max.
  0.000   0.000   1.000   2.262   2.750  30.000
Generalized linear mixed model fit by maximum likelihood (Laplace
Approximation) ['glmerMod']
 Family: poisson  ( log )
Formula: resp.var ~ 1 + Treat_f + (1 | Pair_f)
   Data: data
 Offset: Lhooks

     AIC      BIC   logLik deviance df.resid
   456.1    464.6   -225.1    450.1      123

Scaled residuals:
    Min      1Q  Median      3Q     Max
-2.0012 -0.5335 -0.5101  0.4959  2.3621

Random effects:
 Groups Name        Variance Std.Dev.
 Pair_f (Intercept) 2.068    1.438
Number of obs: 126, groups:  Pair_f, 63

Fixed effects:
             Estimate Std. Error z value Pr(>|z|)
(Intercept) -5.079269   0.241152 -21.063   <2e-16 ***
Treat_fCap  -0.007019   0.118468  -0.059    0.953
```

```
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Correlation of Fixed Effects:
           (Intr)
Treat_fCap -0.245
  Treat_fCap
-0.007018999  0.118468477
 Treat_fCap  Treat_fCap  Treat_fCap
  0.6994424 -25.8496751  21.6477854

Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
     1.057e-02      2.434e-03      7.696e-05      7.696e-05

2. Quantiles for each variable:

    2.5%      25%      50%      75%    97.5%
0.006150 0.008884 0.010483 0.012118 0.015899


Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
     9.474e-03      2.088e-03      6.604e-05      7.481e-05

2. Quantiles for each variable:

    2.5%      25%      50%      75%    97.5%
0.005770 0.008033 0.009335 0.010757 0.013896

NULL

Iterations = 200001:699501
Thinning interval = 500
Number of chains = 1
Sample size per chain = 1000

1. Empirical mean and standard deviation for each variable,
   plus standard error of the mean:

          Mean             SD       Naive SE Time-series SE
        6.3934        27.6577         0.8746         0.8746

2. Quantiles for each variable:

   2.5%     25%     50%     75%   97.5%
-58.258  -7.893  11.044  24.907  49.216

     2.5%       50%     97.5%
-58.25802  11.04394  49.21579
```


m5d.1$Sol[, 3]
m5d.1$Sol[, 4]
m5d.1$Sol[, 1]
m5d.1$Sol[, 2]
Parameter
MCMC sample
0
200
400
600
800
1000
0
-2
-4
-6


50
0
-50
-100
-150
PercReduction.mcmc
0
200
400
600
800
1000
MCMC sample


0.015
0.010
0.005
0.000
Density
-150
-100
-50
0
50
PercReduction.mcmc

Summary and Management Implications

Summary

In this thesis I have attempted to prepare a current conservation assessment of two closely related seabirds, the shy albatross and white-capped albatross, focussing on the prime threat that both face in the marine environment – incidental mortality in commercial fisheries. My approach was to initially review the levels of effort in fisheries known to kill both species and develop an estimate of the global bycatch level. I then developed and fitted population models for both species to evaluate the impact of bycatch on population growth. I also undertook annual population censuses of white-capped albatrosses over an eight years at their major breeding site) to estimate population size and track population trends. Lastly, I complemented these analyses with at-sea experiments to test the efficacy of a mitigation method with the potential to significantly reduce interaction levels in pelagic longline gear, one of the primary gear types that currently poses a threat to both shy and white-capped albatrosses.

I found that the global impact of fisheries on both species was high, and that many thousands of birds were likely being killed each year. The quality of available fishery bycatch and effort data was relatively poor, which made it necessary to extrapolate across fisheries and seasons to develop a bycatch estimate. Although the reliability of resulting estimates were low due to data paucity, in 2007 it was likely that over 8 500 of these albatrosses were being killed annually. Of these estimated deaths, trawl fisheries were responsible for 75%, with longline fisheries responsible for all other mortality. At that time most of these birds were assessed as being killed in South African, Namibian and New Zealand fisheries. Because most adult shy albatrosses are comparatively sedentary and rarely found outside Australian waters, it is primarily juvenile shy albatrosses that regularly encounter fishing fleets known to kill large numbers of albatrosses. In contrast, throughout most of their range juvenile and adult white-capped albatrosses are exposed to fisheries that collectively kill many thousands of these albatrosses each year.

Fisheries effort varies considerably over time in response to changing economic and biological factors, and recent reappraisal of these data (Chapter 3) showed that estimated observed bycatch levels had declined by 60% but were still high (c.3 200). The decline was largely due to rigorous application of mitigation measures in a South Africa trawl fishery (Maree et al. 2014) and updated information on Namibian fisheries previously thought to be impacting 'shy-type' albatrosses (BirdLife 2013a, 2013b). However, inclusion of an allowance for unobserved or 'cryptic' mortality led to annual global estimates ranging between c.7 000-13 400 for both species combined (Chapter 3).

If current levels of bycatch are unsustainable, population trend data should provide a clear indication of the severity of the impacts. Trend data for shy albatross are available from a long-term study at Albatross Island, Australia (Alderman et al. 2011), but not for white-capped albatross. Almost all of the global population of white-capped albatross breeds in the remote sub-Antarctic Auckland Islands, and between 2006 to 2013 I undertook annual counts of breeding white-capped albatrosses using aerial photography (Chapter 2). Based on these counts the mean number of annual breeding pairs in the Auckland Islands during this period was estimated to be 90 141, with annual estimates ranging from 73 838 to 116 025. The high level of inter-annual variance around the count data leads to difficulties in detecting a trend around annual means. It is therefore not surprising that trend analysis showed no clear evidence of change over the eight years of the study and hence no trend in the total population. This is consistent with the long-running shy albatross study site where no trend has been detected over the last 10 and 20 years (Alderman et al. 2011). Continuation of annual monitoring for both species is recommended, particularly for white-capped albatross, to clarify the population status of this species and determine if current levels of fishing mortality are impacting the population.

As no clear picture emerged on fisheries impact from the population estimates, I conducted a Population Viability Analysis for shy and white-capped albatrosses to further evaluate the present and future impacts of fisheries-related mortality. The models developed indicated that the estimated bycatch from global fisheries is unsustainable for both species, with populations for both species halving over 30 years. However, since such a rate of decline has not been observed in population studies of both these species over the last 10-20 years, it is likely that bycatch estimates have been over-estimated. Given the uncertainty around cryptic mortality multipliers, and the extrapolations required to accurately estimate observable bycatch, and considering bycatch data is based on low sample sizes, this is not surprising. Estimates of Potential Biological Removal (PBR – Wade 1998) calculated through use of a range of recovery factors (F_R = 0.1, 0.2, 0.5 and 1) were also modelled as annual bycatch, and those based on high recovery factors (F_R = 0.5 and 1) were also unsustainable (Chapter 3). Only scenarios that modelled bycatch based on PBRs with F_R = 0.1 or F_R = 0.2, as recommended in the literature for threatened species (Dillingham and Fletcher 2008), led to positive population growth when applied to my base model for both species. Based on my modelling I consider that annual PBR values of 400-800 shy albatross and 900-1 700 white-capped albatross should be sustainable bycatch levels for these species.

While application of PBRs to manage bycatch sustainably can be effective in maintaining populations at prescribed levels, the development and implementation of measures to mitigate the bycatch of albatrosses and other seabirds must also be maintained. A range of mitigation measures for reducing the incidental catch of seabirds in trawl and longline fisheries have been developed over the last 20 years (summarised in Lokkeborg 2011; ACAP 2013 a, b, c) but proven and accepted seabird avoidance measures in pelagic fisheries still require substantial improvement. To help address this situation, I conducted an at-sea experiment in South Africa to test the efficacy of a mitigation method known as the Smart Tuna Hook (Chapter 4). The trials were successful and use of the Smart Tuna Hook led to an 81.8% – 91.4% reduction in the bycatch of seabirds in one of the highest-risk fisheries to seabirds in the world. Importantly, there was no detectable detrimental effect on fish catch for any species. In a fishery where the bycatch rate of seabirds exceeded 1 bird/1000 hooks, and where the capture of more than 25 birds by a vessel each season leads to a suspension of fishing activity for that vessel, the Smart Tuna Hook clearly provided a feasible option for pelagic fishers to significantly reduce bycatch albatrosses and other seabirds and hence remain active in the fishery.

The bycatch of shy and white-capped albatrosses occurs over the entire range of both species and at levels that may be impacting population growth. Reducing bycatch in fisheries poses significant challenges for gear technologists and fisheries managers. Finding solutions requires a mix of legislative and political measures to facilitate industry engagement and provide incentives for action, combined with sound science to define problems and provide robust assessments of the impact of bycatch at a species and population level, and to ensure development and implementation of effective mitigation measures.

Management Implications

Emerging from this research are a number of issues that are relevant to the management of commercial fisheries and the conservation of albatrosses and other seabirds that interact with fishing gear.

- Global estimates of bycatch for seabirds and other non-target species are useful for managing the ecological effects of fisheries, but data collection is still woefully inadequate in most fisheries. It is widely accepted that reliance on logbook data alone does not provide an accurate picture of bycatch levels and patterns, necessitating some level of independent assessment, either through the use of observer programmes or electronic (e.g. video) monitoring. While sampling protocols for electronic monitoring are still being developed, it is generally agreed that at least observer coverage of 20% of all fishing effort (hooks set, number of trawls etc) is necessary for accurate estimation of bycatch levels in fisheries. Achieving observer or electronic coverage at this level would go a long way to greatly improving the accuracy of bycatch estimates, particularly if coverage protocols take into account spatial and temporal aspects for each fishery.

- When observer coverage is low, bycatch estimates are typically based on extrapolations. Such extrapolations are not only potentially inaccurate but also misleading, particularly where observer data is not representative. Notwithstanding this, using such data for bycatch estimation is often necessary as it is the only data available. While uncertainty around such estimates will be high, estimates derived in this way are often useful for indicating potential problems and encouraging fishery managers and fishers to implement appropriate management.

- Most published seabird bycatch estimates are based solely on levels of observed and reported bycatch and make no allowance for cryptic mortality. Such an approach is understandable but such estimates are likely to substantially under-estimate fisheries-related mortality (Francis 2012). More work needs to be done on the development of cryptic mortality multipliers that are gear and species specific. The values developed for white-capped albatrosses in New Zealand (2.08 and 8.23 for longline and trawl gears, respectively; Richard and Abraham 2014) are useful starting points, but clearly require refinement, particularly for trawl gear. If multipliers are to assist in more accurately estimating bycatch, they need to be based on good empirical data and be widely accepted by industry and environmental scientists alike.

- Where longitudinal datasets exist for impacted species, assessment of population trends may be useful for indicating the impact of fisheries bycatch levels. However, in many cases these datasets are not available and commencement of suitable monitoring programmes will be unlikely to provide the immediate evidence necessary to trigger management actions. As evidenced in Chapter 2, population trend for white-capped albatross was uncertain even with eight years of count data collected at a cost of c.NZD $500 000. This was due to annual counts exhibiting strong inter-annual fluctuations, a characteristic observed in many long-term data sets for seabirds. Where inter-annual variance is greater than the variance around mean annual counts, detection of a reliable estimate of trend will require many years of data. As such, reliance in trend data to measure the impact of fisheries on a population should generally be considered a blunt tool, and population modelling may provide more useful information in a shorter timeframe.

- Mitigation of seabird bycatch in commercial fisheries still presents challenges to gear technologists and fishery managers alike. Currently no single measure is considered capable of reliably preventing the incidental mortality of seabirds in most pelagic longline fisheries (ACAP 2013a), and use of the most effective measure for trawl fisheries - retention of offal and discards – has not been widely adopted. The suite of options now recommended as best-practice for pelagic longlining, usually a mix of Bird Scaring Lines, night setting of gear and use of weighted swivels, generally impose a level of inconvenience at best, or severely compromise fishing efficiency at worst. Similarly for trawl gear, full offal retention and management is generally not taken up because existing vessels have not been set up to facilitate this, or there is insufficient space on the vessel to hold large quantities of waste material. Offering mitigation options to fishers that cannot be easily implemented or reduce the catch of target species are likely to be met with resistance. The challenge for the future remains to find ways of saving seabirds while catch of target fish species is maintained or improved. Only 'win-win' solutions are likely to readily adopted and widely used.

- New innovations in mitigation need to be developed to cover all fishing operation where bycatch of seabirds is problematic. Solutions such as the Smart Tuna Hook may provide safe and efficient solutions for commercial fisheries to minimise seabird bycatch, but uptake by industry will undoubtedly be determined by the material costs and other fishing economics, which will immediately impact a fishing operation over and above that incurred during 'conventional' fishing operations. As such, the Smart Tuna Hook and some other forms of mitigation may only be viable in high-value fisheries such as those targeting tunas and billfish. They may not be attractive options in many artisanal or other fisheries where product value is low.

- Widespread voluntary adoption of mitigation measures is unlikely in many fisheries without incentives for fishers to take up such measures. This is particularly the case in artisanal and other fisheries dominated by a fishing fleet that is largely owner-operated and where immediate financial imperatives take priority over longer-term industry sustainability. Incentives need to be in the form of potentially significant improvement in fishing efficiency (such as through increased capture of target species, improvement to operational issues such as easy stowing and deployment of gear, reduced labour costs, and improved fuel economy). Educating fishers in the use of mitigation measures in the absence of appropriate incentives and compliance is unlikely to lead to uptake of those measures and a subsequent reduction on seabird mortality levels.

- Developing robust conclusions about the efficacy of mitigation measures requires experimental testing and the use of quantitative methods. Ideally, data would be sourced from designed experiments conducted at sea, where the mitigation measures in question would be deployed head-to-head against a control of no deterrent. In a recent review of mitigation of mitigation measures for marine mammals (Barry Baker, Sheryl Hamilton and Luke Finley, unpublished) there were few examples of mitigation experiments conducted in this way, which made it difficult to draw firm conclusions on their effectiveness. Future efforts to develop and implement mitigation measures for all non-target taxa in fisheries for should seek to embody these experimental principles.

- The future prospect for many seabird populations impacted by fishing remains clouded by uncertainty. Only a thorough understanding of the ecology of a species and the nature of fisheries interactions, coupled with the widespread adoption of appropriate and effective mitigation measures, will give confidence in ensuring their long-term survival.

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